Using Life Cycle Assessment in Agriculture – Methodological considerations of variability and uncertainty in dairy carbon footprints

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Declaration

Declaration:

I hereby declare that I completed the doctoral thesis independently based on the stated resources and aids.

27.08.2018

Dedication

To my family.

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Abstract

Life cycle assessment (LCA) analyses the environmental performance of products and services and has become increasingly important also for the environmental assessment of dairy systems. In order to create consistent results for communication, declaration and comparison, the International Dairy Federation (IDF) provides a guideline for the calculation of product-related greenhouse gas (GHG) emissions in the dairy sector. However, the effects of farm data variability and emission factor uncertainty on the comparability of GHG assessments on the farming level are seldom considered. This thesis aims to fill this gap, by exemplarily identifying the consequences of data and calculation uncertainty within the IDF methodology, and by providing recommendations for improving the calculation procedure.

In the first study, different settings in the definition of energy corrected milk (ECM) and the reference flow were compared in a calculation example based on average farming data. A high bandwidth of the carbon footprint result indicated a severe uncertainty when calculation procedures are not well documented. In order to reflect temporal representativeness, the second case study examined the inter-annual variability of production data from six consecutive milk years in an organic dairy farm in northern Germany and its effect on the estimation of product-related GHG emissions using a detailed material flow model (FARM - Flow Analysis and Resource Management model). A procedure to deal with inter-annual variability was proposed and performed for the farm under study. It was shown that data from at least four years is needed to provide reliable results for that farm. The third study dealt with the demand of the IDF guidelines to use at least Tier 2 in the methodology of the Intergovernmental Panel on Climate Change (IPCC) in order to provide comparable results. Using data from 20 Norwegian dairy farms, the uncertainty of the carbon footprint using Tier 1 of the IPCC guidelines within the FARM model was assessed. From all 190 direct comparisons of two farms in the study, 78 % of the comparisons were significantly different with a relative difference of 8.7 % being enough to establish significance of the difference.

From the three studies it was concluded that existing rules may partly not be precise enough to allow for comparison of farms or farming systems, or partly too strict and thereby hindering the execution of carbon footprint studies.

Zusammenfassung

Mit Ökobilanzen werden Umwelteigenschaften von Produkten und Dienstleistungen analysiert und zunehmend bei der Bewertung von Milchproduktionssystemen eingesetzt. Um konsistente Berichterstattung und Vergleichbarkeit von produktbezogenen Treibhausgasemissionen im Milchsektor zu gewährleisten hat die International Dairy Federation (IDF) Berechnungsgrundlagen publiziert. Allerdings werden die Effekte von Variabilität betrieblicher Kennzahlen und Unsicherheit von Emissionsfaktoren unzureichend betrachtet. Diese Arbeit hat es zum Ziel diese Lücke zu schließen, indem exemplarisch die Auswirkungen von Daten- und Berechnungsunsicherheiten in der IDF-Methodik auf Agrarbetriebsebene betrachtet werden und Empfehlungen zur Verbesserung abgeleitet werden.

In der ersten Studie wurden verschiedene Definitions- und Berechnungsmöglichkeiten des Referenzflusses und der funktionellen Einheit für die Klimabilanz von Milchproduktion verglichen. Eine hohe Bandbreite an möglichen Ergebnissen – bei gleichen Eingangsdaten – ermöglicht eine große Ergebnisunsicherheit. Die Voraussetzungen für zeitliche Repräsentativität wurden in der zweiten Studie untersucht. Über 6 aufeinanderfolgende Jahre wurde auf einem ökologischen Milchviehbetrieb in Norddeutschland der Effekt von zwischenjährlicher Variabilität auf die Klimabilanz mit einem detaillierten Stoffflussmodel (FARM – Flow Analysis and Resource Management Model) analysiert. Dabei zeigte es sich, dass für den untersuchten Betrieb mindestens 4 aufeinanderfolgende Jahre untersucht werden müssen um belastbare Ergebnisse zu erzielen. Die dritte Studie befasst sich mit der Forderung der IDF Richtlinien mindestens ein Stufe 2 Verfahren der Methodik des Intergovernmental Panel on Climate Change (IPCC) zu verwenden. Mit Daten von 20 norwegischen Milchviehbetrieben wurde die Unsicherheit der Klimabilanz auf Basis von Tier 1 Berechnungen bei bodenbürtigen Emissionen mit dem FARM Modell ermittelt. Von allen 190 direkten Vergleichen von zwei Betrieben miteinander waren 78 % signifikant unterschiedlich. Ein relativer Unterschied von 7,8 % des Carbon Footprints reichte aus um Signifikanz herzustellen.

Aus den drei Studien wird geschlossen, dass die existierenden Regeln zur Erstellung von Klimabilanzen von Milchproduktion teilweise zu unpräzise und teilweise zu streng sind, und damit sowohl Erstellung als auch Interpretation von betrieblichen Klimabilanzen in der Milchproduktion erschwert werden.

Schlagwörter: Uncertainty, dairy, organic, milk, LCA, methodology

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Abbreviations

AGDM	Above Ground Dry Matter
BGDM	Below Ground Dry Matter
CV	Coefficient of Variation
DM	Dry Matter
ECM	Energy Corrected Milk
EPD	Environmental Product Declaration
FAO	Food and Agricultural Organization of the United Nations
FARM	Flow Analysis and Resource Management Model
FU	Functional Unit
GE	Gross Energy
GHG	Greenhouse Gas(es)
IDF	International Dairy Federation
IPCC	International Panel on Climate Change
IQR	Inter Quartile Range
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LUC	Land Use Change
N	Nitrogen
REPA	Resource and Environmental Profile Analysis
SDM	Standard Deviation of the Mean
TAN	Total Ammonia Nitrogen
VS	Volatile Solids

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1 Introduction

1.1 Introduction to life cycle assessment

Life cycle assessment (LCA) analyses the environmental performance of products and services throughout their life cycle (Klöpffer & Grahl 2014). The international standards ISO 14040 and ISO 14044 serve as framework for LCA and guidelines for the carrying out of LCA studies (ISO 2008a, ISO 2008b).

One key aspect of LCA is the focus on the entire life cycle, which means LCA accounts for all relevant environmental effects during the lifespan of one product. The life cycle begins with the extraction of raw materials from the environment, e.g. ore mining or oil production, carries on through the production of intermediate materials, the final production, the distribution of the product, the use phase of the product, and finally the end-of-life phase of the product which may consist of recycling and/or landfill.

The second key aspect of LCA is the use of a functional unit (FU). The FU expresses the "quantified performance of a product system for use as a reference unit" (ISO 2008a). This means that for the analysis of a product not the product itself is under investigation but rather its benefit(s) during the use phase. The environmental performance of two products can be compared when their benefits during the use phase are the same. This is achieved by first defining the intended use and then scaling the product so that the desired function is fulfilled. The scaled amount of the product that fulfills the intended use is the reference flow. For example, the function of food could be expressed as MJ metabolizable energy. For 1 MJ metabolizable energy, 0.36 litres of milk with 4.0% fat and 3.4% crude protein are needed (Koesling et al. 2017)[1]. Thus, the functional unit would be 1 MJ of metabolizable energy and the reference flow would be 0.36 litres of milk.

The standard procedure of LCA according to ISO 14040 (ISO 2008a) defines 4 phases of each study (Figure 1). In 'Goal and scope definition', the framework of the study is defined. This includes the definition of the functional unit and the system boundary but also needs to define the intended audience, intended environmental impact categories,

1 Based on the energy correction algorithm from Sjaunja (1991).

the requirements for data quality, and general modelling decisions such as allocation procedure. During the 'Life Cycle Inventory' (LCI) phase, data are collected and verified, the material flow model is constructed, and all material flows (product flows, waste flows, and emissions) are scaled to the reference flow.

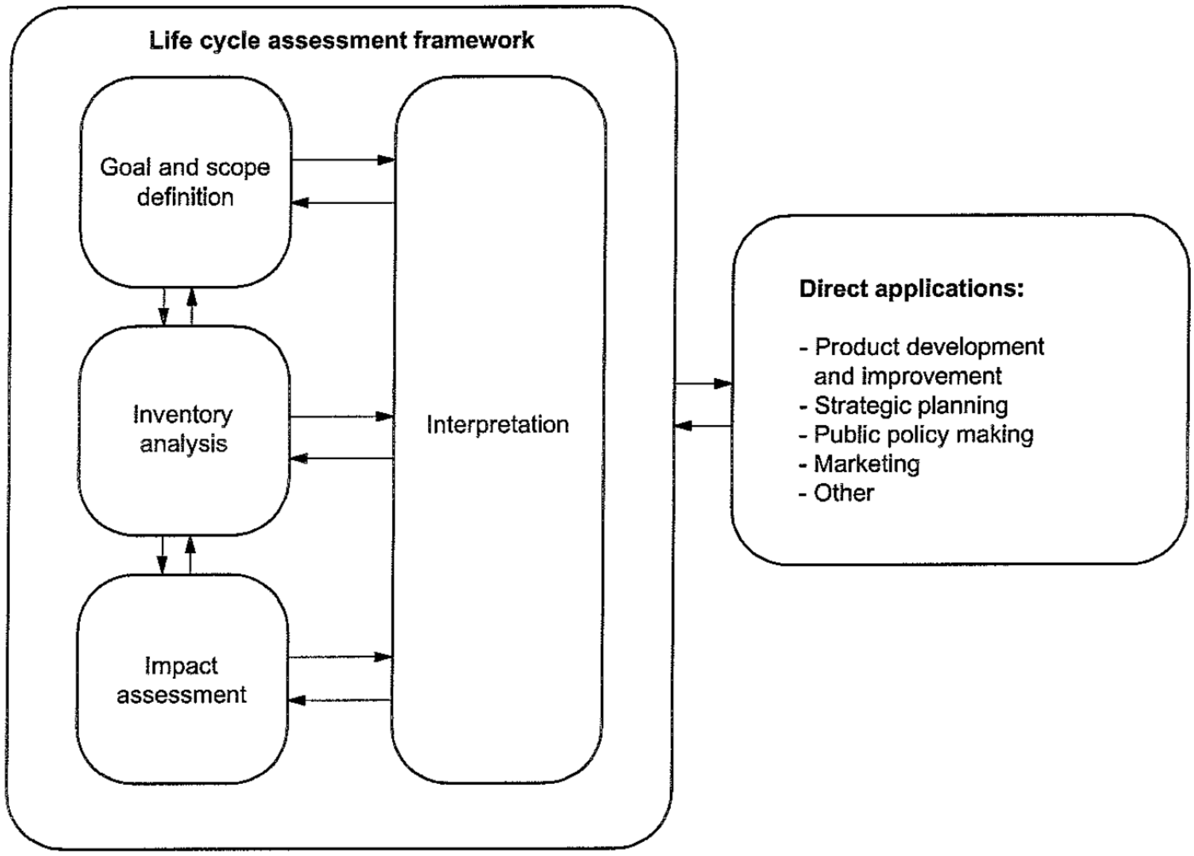

Figure 1: DIN EN ISO 14040:2008 – Figure 1 – Stages of an LCA. Reproduced by permission of DIN Deutsches Institut für Normung e.V. The definitive version for the implementation of this standard is the edition bearing the most recent date of issue, obtainable from Beuth Verlag GmbH, Burggrafenstraße 6, 10787 Berlin, Germany.

The 'Life Cycle Impact Assessment' (LCIA) phase assigns material flows from the LCI to impact categories (e.g. methane emissions to climate change) and multiplies with the characterization factors of each material in the impact category to obtain category indicator results. The set of characterization factors for an impact category is the characterization model.

In the interpretation phase of a LCA, conclusions are drawn from the findings based on a systematic procedure to evaluate the quality of the calculations. It has been found to be inevitable to refine goal and scope, data acquisition, choice of impact categories or other aspects of the study. Hence, it is an inherent quality of LCA to be iterative (Klöpffer & Grahl 2014).

1.2 Life cycle assessment of dairy farming

The beginnings of LCA trace back to the early 1970s, then called resource and environmental profile analysis (REPA) in the U.S.A. (Hunt et al. 1996). These early LCAs often dealt with packaging. The systematic analysis of agricultural production happened as early as 1976 with a report on the energy consumption during the production of agricultural products in the U.K. (Leach 1976).

In the 1990s, more structured LCAs on agricultural products with the inclusion of additional impact categories were performed. These include notably a study on the theoretical basis of the environmental assessment of renewable resources combined with a case study on the environmental performance throughout the life cycle of fuel from rape seed (Reinhardt 1993) for use as fuel. Analyses on the farm level were performed by Wetterich & Haas (1999; Haas et al. 2000) with LCAs on 18 dairy farms with different intensity levels in the Allgäu Region in southern Germany.

During the 2000s, an increasing number of studies on dairy farms were performed dealing with analyses of farming systems as well as with various methodological aspects of LCA on farm level. The analyses of farming systems included the comparison of regions (Flysjö et al. 2011a, Guerci et al. 2013), comparison of organic and non-organic milk production (Cederberg & Mattson 2000, de Boer 2003, Berlin & Uhlin 2004, Thomassen et al. 2008b, van der Werf et al. 2009), and effects of management (Guerci et al. 2014).

Methodological aspects included, but are not limited to, allocation (Cederberg & Stadig 2003, Weidema & Schmidt 2010, Flysjö et al. 2011b, Flysjö et al. 2012), setting of goal and scope, i.e. attributional vs. consequential LCA, (Thomassen et al. 2008a), or uncertainty. Here, emission factor uncertainty (e.g. Chen & Corson 2014), activity data and parameter uncertainty (e.g. Basset-Mens et al. 2009, Wolf et al. 2017, Zehetmeier 2014) and uncertainty from variability (e.g. Guerci et al. 2013,) were in the focus.

All of the above mentioned studies focused on the production of milk on farm and disregarded the life cycle stages after the farm gate. With energy-corrected milk, a quality-corrected functional unit is used (Schau & Fet 2008). This quality correction shall ensure that the reference flows from different systems are in fact equivalent. Equivalence of functions is of very high importance in LCA studies (Klöpffer & Grahl 2014). In some cases the used area is also incorporated into the analysis as a reference flow (e.g. Haas et al. 2000). This is justified by the specificity of the area use in different farming systems. However, in LCA data collections not specific to agriculture such as the ecoinvent LCI database or environmental product declarations (EPD) within the scope of EN 15804, area use is typically considered as a resource input (see e.g. Ecoinvent 2015, EN 15804). It could be argued its use as reference flow is outside of the methodology of LCA.

Recently, some reviews on dairy LCAs have been published, focusing on methodological challenges in practical LCAs (Pirlo 2012) or recent developments in methodology and practice (Yan et al. 2011, Baldini et al. 2017). In all of these reviews, the need for consistency of methods for agricultural LCAs has been stressed. At least since 2010, attempts have been made to harmonize procedures for LCA or carbon footprints on farm level.

A national effort was done in Germany with a calculation standard for carbon footprints on farm level (BEK 2016). This standard is valid for all types of agricultural activity. On international level the International Dairy Federation (IDF) published 'A common carbon footprint approach for the dairy sector: The IDF guide to standard life cycle assessment methodology' in 2010 (IDF 2010). A revised version was published in 2015 (IDF 2015). As part of an environmental product declaration (EPD) system according to ISO 14025 (ISO 2011), environdec (www.environdec.com) has published product category rules for milk. These rules apply when preparing an EPD.

All of these guidelines aim to provide rules that lead to comparable results. However, the variability of agricultural activity across regions, farming systems and even between years is very high. Secondly, activity data and emission data in agriculture are often uncertain. This is especially the case for LCAs with the scope of a practical farm (Schultz et al. 2013). Consequently, the question arises in how far these methodologies for standardization of methods and results deal with variation and uncertainty of data on farm level and of emission factors and what implications these methodologies may have on the comparability of environmental assessments on farm level. This problem is addressed in the current thesis.

The focus in this work is laid on the consequences of uncertainty of data when using the methodology from the IDF. This choice was made as the IDF guidelines deal only with carbon footprints which reduces the complexity of the task. Secondly, it was created alongside internationally accepted methodologies and revised in an ongoing communication with stakeholders, scientists and practicioners (IDF 2015). Lastly, it is published as an international guideline from "a recognized international authority which contributes actively to the development of science-based standards for the dairy sector" (IDF 2017).

Using an accepted methodology for carbon footprints that aims to provide comparable results means that a lot of decisions such as definition of functional unit, system boundaries, and allocation procedures have already been made. This can be a help for the practitioner as the reasoning behind some decisions can be complex or the underlying knowledge not easy to obtain. On the other hand, it can be a hindrance when the approach lacks flexibility to adapt to special situations not foreseen in the creation of the methodology.

1.3 Overview of the IDF guidelines for carbon footprints in the dairy sector

In the context of an LCA-type assessment, the IDF guidelines for carbon footprints correspond to the 'Goal and Scope' phase of LCA. In the following important methodolocial choices that have been made in the IDF guidelines are presented.

The aim of carbon footprints created with the IDF methodology is "[c]omparison of GHG emissions between cattle dairy products, for example 'cheese' or 'liquid milk'" (IDF 2015, p.8). It is not made explicit whether comparisons can only occur within the same study or can also be drawn across different studies, e.g. in a meta-analysis, but it is stated that the guidelines aim to support "consistent and comparable carbon footprint figures” (IDF 2015, p.9) which could indicate that comparisons can indeed be drawn across different studies.

The functional unit is determined as a cattle dairy product at the farm gate or at the factory gate. For on-farm analysis the functional unit is one kilogram of fat- and protein-corrected milk with 4% fat and 3.3% true protein. For processed products it is also recommended to use 1 kg of packaged product as functional unit with a determined fat and protein content (IDF 2015, p. 15).

The system boundaries in farming include production and supply of all inputs to the farm (supplementary feed, fertilizer, etc.) and all production steps and direct emission sources on farm related to milk production. As cut-off rule, materials contributing less than 1% to overall emissions can be excluded when total exclusion does not exceed 5% of total emissions (IDF 2015, pp. 20-21).

Emissions to be included are fossil carbon dioxide (CO_2), biogenic CO_2 from land use change (LUC), fossil and biogenic methane (CH_4), and nitrous oxide (N_2O). Storage of biogenic carbon in packaging should be accounted for while storage of fossil carbon in packaging and emissions in the short biogenic carbon cycle should not be included (IDF 2015, pp. 23-24).

While it is made clear that primary (self-collected) data is preferred, the use of secondary data is not restricted. It is however demanded that the data sources are made transparent and that the temporal, geographical, and technological representativeness is stated (IDF 2015, p. 25).

The IDF acknowledge that under practical circumstances direct emissions from soil and from ruminant digestion cannot be measured but must be estimated. The guideline reference the Tier structure from the International Panel on Climate Change (IPCC) where Tiers 1, 2, and 3 use increasingly detailed methods for the calculation of direct emissions (IPCC 2006, Chapter 1). The IDF guidelines demand to use at least Tier 2 (IDF 2015, p. 26).

Allocation naturally occurs in agricultural systems, e.g., between cereals and straw and between milk and meat. For the allocation in feed production economic allocation is suggested. For allocation between milk and meat a general allocation factor was derived from data that were collected from 536 U.S. dairy farms (IDF 2015, pp. 28-29, 53-55). Allocation for energy generation is performed according to system expansion with credit (IDF 2015, p. 35).

To ensure temporal representativeness of the data, the IDF guidelines demand to use data from at least one year to account for seasonal changes (IDF 2015, p. 25 Footnote 1). This demand is later contradicted in the recommendation to report the three-year average carbon footprint (IDF 2015, p. 41). Furthermore, variation and uncertainty of data should be estimated based a sensitivity analysis or a qualitative discussion (IDF 2015, p. 25).

With the above described framework, the IDF aims to allow the consistent calculation of carbon footprints for use in comparisons as well as the identification of improvement measures (IDF 2015, p.9).

1.4 Goals of the thesis

As described in chapter 1.2 LCA plays an increasing role in the environmental assessment of dairy farming. Chapter 1.3 describes the current documented practice for carbon footprint calculation in the dairy sector. However, drawing experiences from LCA studies in the dairy sector, some rules in the IDF guidelines may need to be reconsidered or challenged. The aim of this thesis is to suggest improvements for the IDF guidelines in regard to uncertainty and comparability of dairy production LCAs. Consequently, this thesis focuses on three methodological issues arising from the IDF guidelines and uses various techniques to tackle these problems. All three issues are written in separate scientific papers in the state of being submitted or published in peer reviewed internationally operating journals.

The first paper (Chapter 2: Schueler & Paulsen 2018c) examines the extent to which lack of transparency hinders the interpretation and comparability of carbon footprints in on-farm assessments. The focus is the definition and calculation of the reference flow in on-farm assessments of cattle milk production. It uses a non-systematic review of scientific publications to point out common problems in the communication of used methods and uses a simple calculation example to quantify the uncertainty from imprecise presentation of results.

The second paper (Chapter 3: Schueler et al. 2018a) aims to improve on the demand of the IDF guidelines to use average data from at least one year to achieve temporal representativeness. While this demand is in line with industry standards for, e.g., the construction sector (EN 15804), its applicability for the agricultural sector is questionable when considering not only seasonal changes but typical inter-annual variation of crop yields, herd sizes, milk yields, etc. The paper analyses variability of farm activity data and its effect on the product carbon footprint of milk of six consecutive years on the research station of the Thünen Institute of Organic Farming in Trenthorst, Germany. Furthermore, a method is proposed to achieve representative results when data is only available for a limited number of years.

The third paper (Chapter 4: Schueler et al. 2018b) aims to show that IPCC Tier 1 methodology can be sufficient in comparative assessments of dairy farming under practical conditions. It refers to the demand of the IDF guidelines to use at least Tier 2 methodology to achieve consistency in dairy LCAs. This demand is challenged with the analysis of 20 Norwegian dairy farms using Monte Carlo simulation for IPCC Tier 1 emission factors for managed soils. With the comparison indicator a method is used that allows determining whether differences between carbon footprint results are significant. The aim of the paper is to explore whether, given a sufficient difference, the variation between farms is higher than the uncertainty induced by IPCC Tier 1.

In the discussion section of this work the findings of the articles are summarized and the importance of uncertainty assessment for dairy LCAs is stressed. Guidance for the practical comparison of dairy LCA results in on-farm assessments is proposed. This guidance is recommended to be integrated into the IDF guidelines to ensure that results obtained with these guidelines can in fact be used for comparisons of product-related GHG emissions in the dairy sector.

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2 Effect of Energy Correction on LCA

Landbauforschung

Title: Effect of choice of reference flow and energy correction formulas on results in life cycle assessment in dairy production

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Zusammenfassung

Ökobilanz (Life cycle assessment, LCA) wird für die Beurteilung der ökologischen Nachhaltigkeit von Milchviehsystemen immer wichtiger. Obwohl Ansätze zur Standardisierung vorliegen erfolgt die Definition von Funktioneller Einheit und Referenzfluss unterschiedlich, auch wenn diese jeweils als Energiekorrigierte Milch (ECM) angegeben werden. Der Referenzfluss sollte die Milchmenge am Hoftor sein, um Verluste und Kälberfütterung mit einzubeziehen. Die Energiekorrektur von Rohmilch besteht aus der Berechnung des Energiegehaltes der Rohmilch und der Skalierung auf den Energiegehalt von ECM. Während die Formeln zur Energiegehaltsberechnung nur wenig voneinander abweichen, existiert kein Konsens über den Energiegehalt von ECM. Dies ist in allen Fällen eine willkürliche Festlegung. Die Futteraufnahme auf Basis der Milchleistung zu berechnen ist ebenfalls abhängig von der ECM Berechnung. Verschiedene Energiebedarfe für dieselbe Menge ECM kann zu unterschiedlichen Futteraufnahmen führen und daraus folgend zu unterschiedlichen Bewertungen der Ressourceneffizienz und Umweltauswirkungen. Werden also keine Informationen zu Berechnung und Definition von Referenzfluss und ECM gegeben, unterliegen LCA Ergebnisse einer großen Unsicherheit. Wir haben verschiedene Parameterkombinationen in einer Beispielrechnung für den Carbon Footprint von Milchproduktion untersucht und eine Unsicherheit von 33% der Ergebnisse gefunden. Um sinnvolle und vergleichbare LCA Ergebnisse zu produzieren, müssen die Definition und die Berechnung von Referenzfluss und funktioneller Einheit transparent dargestellt werden.

Abstract

Life cycle assessment (LCA) is increasingly important for the environmental assessment of dairy systems. While efforts to standardize procedures are being made, many studies define the functional unit and reference flow in a different way even though they all refer to energy corrected milk (ECM). The reference flow should be the amount of ECM at the farm gate to account for losses and milk fed to calves. The calculation of raw milk to ECM consists of the calculation of energy of raw milk and the scaling to the energy content of ECM. While the different formulas to calculate the energy content of raw milk differ only slightly, no consensus exists on the energy content of ECM, as it has been an arbitrary choice in all instances. Calculating the feed demand based on milk yield is also sensitive to the ECM calculation. Different energy demands for the same amount of ECM can lead to different calculated feed intakes, and consequently different resource efficiencies and environmental impacts. Consequently, when no information on the definition and calculation procedure of ECM is given, LCA results may face a severe uncertainty. We evaluated the effects of different settings on carbon footprint of milk in a calculation example and found an uncertainty of 33% to either side of the results. In order to provide valid LCA results, the definition and calculation procedure of the functional unit and reference flow must be transparently disclosed.

Key words: Ökobilanz, EKM, Vergleichbarkeit, agrar, Milch

2.1 Introduction

Life Cycle Assessment (LCA) plays an increasing role when assessing the environmental performance of dairy production (Baldini et al. 2017). However, LCA results may face acceptance problems due to high uncertainty or lacking trust in the uncertainty assessment (Herrmann et al. 2014). Many studies tackle various aspects of uncertainty when assessing carbon footprints. These aspects include emission factor uncertainty (Chen & Corson 2014, Schueler et al. 2018a), activity data and parameter uncertainty (Basset-Mens et al. 2009b, Wolf et al. 2017, Zehetmeier 2014), or spatial or temporal variability (e.g. Guerci et al. 2013, Schueler et al. 2018b).

With ‘A common carbon footprint approach for the dairy sector’ from the International Dairy Federation (IDF 2015) and the product category rules for raw milk according to ISO 14025 from International EPD® System (www.environdec.com) two guidelines exist that aim to produce reproducible and comparable results and stress their relationship to the LCA norm 14040 (ISO 2008). While the IDF guidelines have notably been used for carbon footprinting in the dairy sector (e.g. Dalgaard et al. 2014, Daneshi et al. 2014, Gollnow et al. 2014, and Jayasundara and Wagner-Riddle 2014), both guidelines are not binding.

A common scope for carbon footprinting is the cradle-to-farm gate analysis where the functional unit is defined as “1 kg energy-corrected milk (ECM)”. Differences in the definition and calculation of ECM have been found in Baldini et al. (2017) and Yan et al. (2011) but dismissed as “slightly different” (Baldini et al. 2017). Of the two guidelines, IDF demands energy-corrected milk (as “fat and protein-corrected milk”) while the International EPD® System obtains carbon footprints per kg raw milk.

Our hypothesis is that definition and calculation of ECM as functional unit is an important source of uncertainty in LCA. We test this hypothesis by showing that uncertainty induced by definition and calculation of ECM results in relevant differences in carbon footprint of milk when assessed with different approaches.

2.2 Material and Methods

In the following, we will address three problems that arise when using ECM as a functional unit and might influence the results:

- **Reference flow definition**
- **Reference flow calculation**
- **Calculation of feed intake based on produced ECM**

To check the effects different modelling choices or algorithm choices might have in carbon footprinting of milk, we used average data from 35 dairy farms from a network of organic and non-organic dairy farms in Germany (www.pilotbetriebe.de; Hülsbergen and Rahmann 2014). The average number of cows in 2015 was 102 with 7,376 kg raw milk produced per cow. Average fat content was 3.83 % and average crude protein content was 3.37 %. These values are based on monthly milk control data, assessing each cow. Of this milk, on average only 6,169 kg were delivered, which includes private use and direct marketing. The remaining production had either been fed to calves or had been discarded due to retention periods.

For the sake of simplicity, we assumed yearly greenhouse gas (GHG) emissions of 1,200 tons CO2-equivalents for the entire dairy system of which 1,000 tons CO2-eq (83 %) are allocated to milk. This leaves 1.59 kg CO2-eq per kg delivered raw milk. Comparable carbon footprints of milk production are also reported in studies of Pirlo (2012) or Guerci et al. (2013).

2.3 Results and discussion

Reference flow definition

According to ISO 14040:2006 the reference flow in LCA is defined as ‘measure of the outputs from processes in a given product system required to fulfil the function expressed by the functional unit’ (ISO 2008).

This definition of the reference flow, in which the *output of a product system* is used as a measure, is not ambiguous. Nonetheless, in practical use two basic options have emerged and have been used for definition of the reference flow in cradle-to-farm gate assessments: the produced amount of milk (e.g. Basset-Mens et al. 2009b, Haas et al. 2001) or the delivered amount of milk (e.g. Castanheira et al. 2010, Cederberg & Mattson 2000, Thomassen et al. 2008, van der Werf et al. 2009).

In some reports, the choice is unclear (e.g. Casey & Holden 2006, del Prado et al. 2010, Schils et al. 2006). In case the delivered milk is defined as ‘sold milk’ it is still possible that private use, e.g. for direct selling or own processing, remains unaccounted for. We suggest clarifying that the functional unit includes both, sold milk and private use.

Reference flow calculation

When ECM is chosen as reference flow, the output of raw milk is scaled to the energy content of ECM. The scaling factor - that is multiplied with the amount of raw milk - therefore comprises two elements: the energy content of the raw milk and the energy content of ECM. The generalized algorithm for the correction formula is found in Gaines (1928):

Formula (1) $kgECM = kgrawmilk \times \frac{Energycontent_{rawmilk}}{Energycontent_{ECM}}$

For the calculation of the energy content of milk, various algorithms exist that take various components of raw milk into account. The energy content of ECM can be expressed explicitly with a unit of energy or implicitly with appropriate fat and crude protein contents. When fat and protein contents are given, these have to be inserted into an appropriate algorithm to obtain the corresponding energy content.

To compare the effect of choice of algorithm we chose four different energy calculation formulas that are frequently used. We only considered algorithms that have coefficients for both, fat and protein content of raw milk and a linear factor for all other components (Table 1). The general form of these algorithms is:

Formula (2) $Energycontent_{rawmilk} = x1 \times fat\% + x2 \times crudeprotein\% + x3$

Table 1: Coefficients for the calculation of energy content of milk adapted to the generalized form and metric units

Source	Fat coefficient x1	Crude protein coefficient x2	Linear factor X3
Sjaunja et al. (1991)	0.383	0.242	0.783
Clark et al. (2001)	0.389	0.229	0.803
GfE (2001)	0.38	0.21	1.05
Tyrell & Reid (1965)	0.376	0.209	0.948

The formula from Sjaunja et al. (1991) was used numerous times in LCA and carbon footprint studies involving Scandinavian countries (Yan et al. 2011). The formula from Clark et al. (2001) is the basis for ECM calculation in the IDF guidelines (IDF 2015). However, in the IDF guidelines true protein is used instead of crude protein and the energy content was included implicitly into the formula, i.e. the factors x1, x2, and x3 were divided by 0.7576 Mcal kg^{-1} ECM, which is equivalent to 3.172 MJ kg^{-1} ECM. The formula from GfE (2001) is used for the German milk control system and forms the basis for feed demand calculations of dairy cows in Germany. The formula from Tyrell and Reid (1965) has been used frequently for the evaluation of feeding strategies in the Journal of Dairy Science (e.g. Bernard & Calhoun 1997, Boyd et al. 2013).

The amount of energy in ECM has been an arbitrary choice in all correction formulas (see formula 1) used by the different authors. Sjaunja et al. (1991) justify their choice of 3.14 MJ as being the average value of other published values. The IDF guidelines provide no rationale for the choice of 0.7576 Mcal (4% fat and 3.5% crude protein, 3.17 MJ). Similarily, GfE (2001) does not justify the choice of 3.28 MJ (4% fat and 3.4% crude protein) as standard, whereas Tyrell and Reid (1965) chose 3.14 MJ kg^{-1} ECM (340 kcal $pound^{-1}$ ECM) to reflect a fat content of 4% as introduced by Gaines (1928). Nonetheless, the energy prediction formula of Tyrell and Reid (1965) is also often used in conjunction with fat and crude protein contents of 3.5% and 3.2%, respectively. Examples are Bernard & Calhoun (1997), Boyd et al. (2013), and Pagani et al. (2016). This would mean 1 kg ECM contains 2.86 MJ (Formula (2) with coefficients from Table 1). These contents are the pricing standard for milk in the United States of America (Neil Michael, Arm and Hammer Animal Nutrition, Princeton, NJ, personal communication and Jerry Cessna, Economic Research Service, USDA, personal communication), which appears to be the reason for this choice in the studies mentioned above.

With the different formulas, we calculated the energy content and the amount of ECM for milk with the different fat and protein contents from the sources and the average values from the German pilot farms (Table 2). For comparability, we changed units to SI units.

Table 2: Energy contents (MJ) and scaling factors (kg ECM) for raw milk to ECM resulting from of the different energy correction formulas and different fat and protein contents according to different standards.

Settings		Calculation results							
Fat	**Crude Potein**	**Sjaunja (1991)**		**Clark et al. (2001)**		**GfE (2001)**		**Tyrell & Reid (1965)**	
%	%	MJ	kg ECM	MJ	kg ECM[1]	MJ	kg ECM	MJ	kg ECM[2]
3.5	3.2	2.90	0.92	2.90	0.91	3.05	0.93	**2.93**	**1.00**
3.83	3.37	3.07	0.98	3.06	0.97	3.21	0.98	3.09	1.05
4.0	3.3	3.11	0.99	3.11	0.98	3.26	0.99	3.14	1.07
4.0	3.4	**3.14**	**1.00**	3.14	0.99	**3.28**	**1.00**	3.16	1.08
4.0	3.5	3.16	1.01	**3.17**	**1.00**	3.31	1.01	3.18	1.09

[1] Assuming 4.0% fat and 3.5% crude protein
[2] Assuming 3.5% fat and 3.2% crude protein

We found that the energy contents we calculated with the different standards are very similar at the same protein and fat contents, except for the results gained with the coefficients of GfE (2001). As stated above, they refer to feed energy demand per kg ECM and consequently calculations ended up in higher results. Subtracting 0.1 MJ difference between energy content and energy demand (GfE 2001) would close this gap to ~1%.

Using a different energy content of 1 kg ECM led to larger differences. Assuming 2.93 MJ kg^{-1} ECM (3.5% fat and 3.2% crude protein, according to Tyrell & Reid (1965)) yielded up to 9% more ECM than assuming 4.0% fat and 3.5%, increasing with increasing protein content.

When different assumptions of fat and protein content of standard milk would be made, the ECM scaling may be even further off. For example, Rotz et al. (2010) assumed 3.5% fat and 3.1% protein without disclosing whether they mean crude protein or true protein.

To summarize, the choice of energy calculation formula is not an important source of differences but the choice of energy content (fat and protein contents) is very important. Consequently, when neither energy content nor fat and protein contents of ECM are disclosed, the uncertainty of results will be very high.

Calculation of feed intake

In most LCA studies, feed intake of the cattle is a very important factor and will influence the results on environmental performance. The feed intake can be calculated based on the energy demands for metabolism and production (e.g. Flysjo et al. 2011, Jayasundara and Wagner-Riddle 2014). Typically, the offered amount of some feed components and their quality are known. The quality of others, as well as the actual intake of most components are unknown. As common approach, the energy supplied by known feed components is subtracted from the feed energy demand for metabolism, live mass increase and milk production to estimate the uptake of unknown components of the ration. Consequently, any uncertainty of total energy demand has a direct impact on the estimation of the uptake of unknown components. As an example, we assume that the difference between well-known feed uptake in form of feed conserves (roughage and concentrates) and total available feed is 10%. These 10% are taken in by grazing. Increasing the total feed demand of cattle 5% with constant feed offer by the feed conserves, would increase the calculated grazing intake by 50% This could affect the assessment of resource efficiency of pastures within a given system.

As described, depending on the availability of data, the feed demand may also serve to calculate other feed components. Then a higher estimate of feed demand could lead to higher estimations of resource use and associated emissions in the process chain of production on farms

So, during crop production, when using an IPCC Tier 1-type approach (IPCC 2006) for emission calculation, higher feed demand would also lead to higher estimates in yields, and consequently in crop residues and increased associated N_2O-emissions. This is also valid for higher than Tier 1 approaches for the calculation of greenhouse gas emissions during crop production when they are sensitive to crop yields (e.g., Bouwman et al. 2002). This means that, just as with IPCC Tier 1, a higher yield calculated from a higher feed demand leads to an increase in calculated N_2O-emissions from crop residues. In addition, Tier 2 or Tier 3-type approaches for estimation of methane emissions from enteric fermentation of cattle may lead to higher values, when feed demand changes by model settings. In short, the estimation of the feed demand may have significant effects on the results of a milk carbon footprint.

When calculating feed demand from different ECM formulas, different assumptions of energy content for the same amount of ECM can occur. For 4.0 % fat and 3.4 % protein both Sjaunja et al. (1991) and GfE (2001) assume 1 kg of ECM (Table 2). However, the energy content of Sjaunja et al. (1991) is 3.14 MJ while GFE assumes 3.28 MJ. The reason is that GfE (2001) distinguishes between energy content of milk (3.18 MJ kg^{-1} ECM) and feed energy demand for the same amount of milk (3.28 MJ feed demand kg^{-1} ECM) while Sjaunja et al. (1991) claim that 3.14 MJ kg^{-1} ECM 'seems to be accepted for application for feeding purposes'. In return, this means that for the same amount of milk with 4.0% fat and 3.4% protein Sjaunja et al. (1991) accept 3.14 MJ energy requirement while GfE (2001) assume an energy requirement of 3.28 MJ. This is an increase of 4.5%, which may have the system-wide effects described above.

Calculation example

We calculated the carbon footprint of our simple example (milk with 3.37 % fat, 3.07 % crude protein) with different reference flow definitions (milk delivered, milk produced) and with the different energy contents for ECM as given in Table 2 resulting from the different formulas. The lowest energy content of 2.86 MJ kg^{-1} ECM produced the lowest carbon footprint in this comparison when produced milk is addressed (Table 3). Whereas sold milk with 3.17 MJ kg^{-1} ECM had the highest carbon footprint. That means that for the same dairy system we could arrive at values between 1.23 kg CO_2-eq kg^{-1} ECM and 1.64 kg CO_2-eq kg^{-1} ECM, i.e. a difference of 33% just from the different definitions of the reference flow. Of these, around two thirds come from the definition of the reference flow and one third from the energy content of ECM. As mentioned above the different parameters for the energy content calculation lead to very similar results.

Table 3: Calculation example for the effect of choices of reference flow and energy correction formula with different assumptions of energy content in ECM

Reference flow	Unit	Sjaunja (1991) 1 kg ECM ≍ 3.14 MJ	Clark et al. (2001) 1 kg ECM ≍ 3.17 MJ	Tyrell & Reid (1965) 1 kg ECM ≍ 2.86 MJ	No correction 1 kg raw milk
1) Produced milk 7.376 kg cow^{-1} yr^{-1}	kg CO_2-eq kg^{-1} ECM	1.36	1.38	1.23	1.33
2) Sold milk 6.169 kg cow^{-1} yr^{-1}	kg CO_2-eq kg^{-1} ECM	1.63	1.64	1.47	1.59

This difference directly translates into results' uncertainty. When identical results from two studies are given without any context on the definition and calculation of the reference flow or calculation, one of the systems could have 33% higher product-related GHG emissions than the other. Vica versa, dairy systems with similar environmental performance could be judged to be far apart, due to lack of transparency in the calculation process.

This uncertainty does not apply, when two different systems are compared within the same study. Multiple studies could consistently find relevant differences between different farming strategies, e.g. organic versus non-organic farming. However, when comparing different results across different studies, e.g. for deriving regional differences, the scaling of the functional unit may lead to false conclusions.

2.4 Conclusion

The method of scaling to the reference flow does not prohibit improving the understanding of a production system, as can be an aim of LCA (Hellweg & Canals 2014). However, when the aim is to provide results for use in comparative assertions, the scaling to the reference may significantly alter the interpretation. Hence, it is of upmost importance to provide a high transparency on the methods and data and not assume terms such as ECM to be sufficiently self-explanatory.

We suggest defining the functional unit and reference flows as follows: “The functional unit is 1 kg energy corrected milk (ECM) at the farm gate (including private use, if applicable). The energy correction is performed using the formula given by IDF (2015) and scales to 3.17 MJ per kg ECM.”

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3 Accounting for inter-annual variability of farm activity data for calculation of greenhouse gas emissions in dairy farming

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Keywords: LCA, milk, organic farming

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Abstract

Purpose

This study examines the inter-annual variability of production data in an organic dairy farm and its effect on the estimation of product-related greenhouse gas emissions (GHG) using a detailed material flow model. It is believed that the examination of only one production year may not adequately reflect temporal representativeness, and may therefore lead to unreliable results. The current study also provides a method to deal with variability when temporal representativeness cannot be ensured.

Material and Methods

All material flows related to milk production from six consecutive milk years in an organic dairy farm in northern Germany were analysed. The milk yield of the 75 to 91 cows varied between 5418 and 7102 kg energy corrected milk per cow and year. GHG emissions were estimated using calculation guidelines from the International Dairy Federation (IDF) and the Intergovernmental Panel on Climate Change (IPCC). Emissions were calculated in the FARM (Flow Analysis and Resource Management) model ensuring mass balances for nitrogen and phosphorous in every subsection of the model. Based on the variability of crop yields, the number of years for representative average data was calculated as well as an uncertainty when only a limited number of years was available.

Results

Estimated GHG emissions varied between 0.88 and 1.09 kg CO_2-eq kg^{-1} ECM^{-1} (mean, standard deviation of the mean: 0.97 and 0.07 kg CO_2-eq kg^{-1} ECM^{-1}). Emissions from ruminant digestion had the highest contribution (50.9±2.3) % in relation to overall product related GHG emissions. Direct emissions from soil showed the highest coefficient of variation (36 %) due to simultaneous changes in fertilization amount, crop yield and milk yield which showed no significant direct relationship. The number of years needed to be assessed for representative average yields was between 27 and 215 years for clover grass and maize silage, respectively. When performing a

sensitivity analysis based on the variability of crop yields, the assessed farm showed reliable results with average data of at least four years.

Conclusions + recommendations

Temporal representativeness should be dealt with explicitly in GHG assessments for dairy farming. If the representativeness of crop yields cannot be ensured, an uncertainty bandwidth of the results based on variability of yields can provide a basis for comparing different farms or farming systems. This approach could also be extended to other variabilities in dairy farming for more reliability of results.

3.1 Introduction

Common agricultural policy in the European Union support beneficial environmental performance by including greening components in the calculation of direct payments (EC 2009). Life cycle assessment (LCA) could play an integral role in quantifying environmental effects serving as one tool to calculate sustainability efforts of farms. To fill that role, methods of calculation should be clear and results must both be reliable and comparable.

The International Dairy Federation (IDF 2010) provides a guideline for the calculation of product related greenhouse gas (GHG) emissions in the dairy sector. The aim of this guideline is to produce consistent results for communication, declaration and comparison. While the uncertainty of emission factors is addressed specifically and a sensitivity analysis is suggested, the guideline does not require averaging data from multiple years to gain representative results. However, the temporal representativeness of the underlying data should be considered in any LCA (ILCD 2010). As the natural variability of processes between years is one of the major differences of agricultural production compared with industrial processes, the effect of inter-annual variation should be known, especially when comparing farms or farming systems.

Many LCA studies on milk production assessed only one production year or calendar year (Thomassen et al. 2008a; Thomassen et al. 2008b; Guerci et al. 2013; Guerci et al. 2014; Thoma et al. 2013; Cederberg and Mattson 2000; Haas et al. 2000). More than one year were assessed in Müller-Lindenlauf et al. (2010) where an average of 6 years was used. To the best of our knowledge no LCA study exists analysing the variability of environmental performance within one farm across several years.

The objective of the study is to assess the influence of production data and their inter-annual variation in an organic dairy farm on product related GHG emissions of milk based on the IDF guidelines (IDF 2010). Furthermore, an approach is developed to assess uncertainty of calculation results induced by inter-annual changes of farm activity data in crop production. An organic dairy farm in Northern Germany is used as a detailed example for calculation.

3.2 Material and methods

3.2.1 Farm under study

The dairy farm of the Thünen Institute of Organic Farming, Trenthorst, Germany, was analysed. It is certified organic according to Council Regulation No 2092/91 (EC 2007). Two dairy breeds – Red Holstein Double Usage and Holstein-Frisian – were kept under organic management conditions. For this study all material flows in relation to the dairy cattle have been analysed for the years 2007 till 2012. Basic conditions of the farm and its management and model parameters were as follows.

All feedstuff, including the concentrates, was produced entirely on site. Only mineral additions and lime were imported. No phosphorous or potassium fertilizers have been imported on the farm since 2001. For the dairy cattle, including their offspring, 62 ha of arable land and 54 ha of permanent grassland were available. The arable land was divided into 6 fields of roughly the same size to allow for a 6-year crop rotation (this is: clover grass, clover grass, silage maize, winter wheat, oat/field beans mixture, triticale). These 116 ha marked the spatial system boundary in this study. Clover-grass and maize were harvested by a chopper. Silages were stored in bunker silos. Yields from crop production were weighed before entering the storage silos. Yields for the years 2006 – 2011 are presented in Table 4. Total nitrogen inputs from slurry and farm yard manure are presented in Table 5. If available feed was not sufficient, feed from other parts of the farm under the same management was used. These other parts of the farm cover separate different production systems in organic farming such as piglet production, dairy goat husbandry and organic cash crop production.

Table 4: Yields from crop production in t dry matter ha^{-1} yr^{-1} and coefficient of variation (CV) for the years 2006-2013 at the research station in Trenthorst/Wulmenau

	2006	2007	2008	2009	2010	2011	2012[a]	2013[a]	Mean	CV
Clover/ grass	6.50	7.80	8.05	8.14	7.04	5.73	8.89	7.48	7.45	0.126
Maize	4.00	9.94	8.87	9.77	5.17	11.4	15.3	7.37	8.98	0.372
Wheat	2.49	4.24	2.83	2.11	3.00	4.10	2.48	3.17	3.05	0.235
Triticale	4.33	3.78	2.39	2.84	2.20	4.30	2.51	3.19	3.19	0.250
Oat/ field bean	4.48	4.17	2.78	2.30	3.20	3.70	4.21	2.67	3.29	0.213

[a] Yield data added only for the calculation of the variability

Table 5: Total nitrogen (N) inputs from slurry and farm yard manure to crops in kg N ha^{-1} yr^{-1} for the years 2006-2011 at the research station in Trenthorst/Wulmenau.

	2006	2007	2008	2009	2010	2011	Mean
Clover/ grass	67	72	0	110	43	30	54
Maize	0	0	0	178	44	6	38
Wheat	135	135	105	141	79	29	104
Triticale	115	394	190	148	43	166	176
Oat/field bean	90	230	0	124	0	132	96

The number of dairy cows kept on farm ranged between 74.5 and 91 (Table 6). All cows were held in a loose housing stable with straw bedding in the cubicles. The cows had access to pasture since 2009. During the grazing period in between April and October they were on pasture during the day and had access to concentrate feed and additional roughage in the stable overnight. Young stock and heifers were held in a separate building. During grazing periods, the latter groups spent 24 hours per day on the grassland.

Table 6: Average number of animals in feeding group, milk yield per cow in energy corrected milk (ECM) and live mass increase of all feeding groups for the years 2007-2012 at the research station in Trenthorst/Wulmenau

Typ	Unit	2007	2008	2009	2010	2011	2012
Suckling calves	|	15.3	12.4	12.9	16.7	16.2	11.9
Calves	|	11.1	10.5	10.9	10.7	10.6	10.6
Young stock	n	33.9	35.2	38.7	41.7	37.1	27.7
Heifers	|	22.3	29.6	25.8	27.3	28.6	23.1
Cows	|	74.5	78.0	86.9	86.8	91.0	82.0
Milk yield	[kg ECM cow^{-1} a^{-1}]	5418	6106	6934	6311	5865	7102

Live mass increase	[kg a^{-1}]	24470	33958	42292	37267	38241	22270

The average milk yield between 2007 and 2012 was 6289 kg energy corrected milk (ECM) cow^{-1} a^{-1}. Milk production data was taken from monthly milk control records. Live mass increase of the animals was calculated by regularly weighing each animal and averaging the daily increase for each feeding group. The live mass increase did not necessarily correspond to meat sold in the same year since the herd structure (number and age of the animals) varied. Animal numbers, milk yields, and live mass increases are shown in Table 6.

All roughages in the stables were offered together as mixed ration. The cattle were divided into six categories and combined in four feeding groups provided with different diets: calves and suckling calves, young stock, heifers, and cows (lactating and dry cows are one group). The definitions of the categories are given in Table 7. Each day the total fodder amount for these groups was weighed before entering the stable. To ensure consistency, fodder flows were scaled to reflect dry matter contents of 35 %, 30 %, and 88 % for maize silage, clover grass silage, and concentrates, respectively.

Table 7: Animal groups and their definitions used in the FARM-Model

Category	Definition
Suckling calves	Calves from 0 days until 90 days with access to whole milk
Calves	Calves from 91 days until 180 days
Young stock	Animals from 181 days to the first insemination
Heifers	Animals from first insemination to first calving
Lactating cows	Cows during lactation period
Dry cows	Cows between two lactations

The actual roughage intake of the animals was not known, as leftovers were not weighed. Instead, leftovers from the cows were brought directly to young stock and heifers. This made the determination of the fodder uptake of each feeding group uncertain. This uncertainty has also been found in Schulz et al. (2013) and seems inherent to using practical farm data. To deal with this, the feed intake of young stock and heifers was estimated based on their energy requirements according to Jeroch et al. (1999). So, total fodder entering the stable, minus feed for young stock and heifers, was assumed to be consumed by the cows and feed losses in the stable are not accounted for. This may lead to an overestimation of feed intake by the cows but represents the complete material flow in a farm view. Concentrate feed was provided through feed dispensers for each cow based on individual milk yield. In this way, the daily concentrate intake for the lactating cows was known.

Even though suckling calves received up to 8 litres whole milk per day and calf, the same amount of roughage and concentrate intake as for the calves was assumed for them, as no reliable distinction was possible. We think that this overestimation is acceptable, as the effect is minor in relation to overall results. Also, it is necessary to provide roughage to suckling calves irrespective of the nutritional value for them to allow development of the rumen functionality (Meyer 2005). Furthermore, due to hygienic reasons, fresh feeds are offered regularly and leftovers are taken out. Consequently, the calculated intake of calves and suckling calves varies very much. The energy requirement for calves between 0 and 180 days averages a feed intake above 2 kg dry matter animal^{-1} d^{-1} (Jeroch et al. 1999). Care was taken as to not underestimate feed intake of the calves.

While it had been shown on the research farm that daily dry matter intake during grazing varies between plots (Ohm et al. 2014), the intake of grass during grazing was estimated based on grazing time. This was necessary as data on daily dry matter intake with grazing were not available consistently over the entire period. An average dry matter intake of 7.0 kg d^{-1} cow^{-1} was assumed for part time grazing (Jeroch 1999). For the full day grazing of young stock and heifers, the calculated energy demand was used as basis for dry matter intake. The diets for each feeding group which were derived are presented in Table 8.

All slurry from all feeding groups, wastewater from the cleaning cycle of the milking parlour and rainwater around the stables were collected in two slurry tanks each holding 2500 m^3. The slurry developed a natural crust. It was not stirred or homogenized during storage; this was done only prior to application. The amount of slurry produced as basis for emissions from manure storage was calculated based on dry matter intake of the cattle according to Windisch et al. (1991). The amount of manure that was applied to the crops can differ from the amount that has been produced as the storage serves as a buffer.

Table 8: Average feed composition given in dry matter intake for the years 2007-2012 at the research station in Trenthorst/Wulmenau for lactating cows (CM), dry cows (CD), young stock (JS), heifers (HE), suckling calves (SCA), and calves (CA)

Feeding group	Type	DM content [kg kg^{-1}]	Daily dry matter intake [kg DM d^{-1}]						
			2007	**2008**	**2009**	**2010**	**2011**	**2012**	**Mean**
Cows	Concentrate (lactating)	0.88	3.7	4.2	4.6	4.0	4.4	4.5	4.23
	Concentrate (dry cows)	0.88	1.1	1.1	1.1	0.8	1.1	1.2	1.06
	Grass silage	0.35	10.8	12.5	11.9	9.4	13.2	9.5	11.3
	Maize silage	0.35	1.6	2.0	2.1	2.7	2.0	5.8	2.7
	Grazing	0.18	-	-	2.5	2.6	2.4	3.0	1.75
	Total		**16.1**	**18.7**	**21.1**	**18.7**	**22.0**	**22.8**	
Young stock and heifers	Concentrate	0.88	0.1	0.4	0.0	0.3	0.4	0.3	0.26
	Grass silage	0.35	3.6	4.6	3.9	2.9	3.2	2.1	3.30
	Maize silage	0.35	0.3	0.6	0.5	0.7	0.3	1.0	0.58
	Grazing	0.18	0	0	1.5	2.0	1.8	4.2	1.58
	Total		**4.0**	**5.6**	**5.9**	**5.9**	**5.7**	**7.6**	
Suckling calves	Concentrate	0.88	0.6	1.4	2.1	1.0	0.9	0.9	1.17
	Grass silage	0.35	2.0	1.8	1.8	0.7	1.1	1.1	1.28
	Milk	0.13	0.6	0.9	1.1	0.7	0.5	0.5	0.73
	Total		**3.2**	**4.1**	**5.0**	**2.4**	**2.5**	**2.5**	
Calves	Concentrate	0.88	0.6	1.4	2.1	1.0	0.9	0.9	1.17
	Grass silage	0.35	2.0	1.8	1.8	0.7	1.1	1.1	1.28
	Total		**2.6**	**3.2**	**3.9**	**1.7**	**2.0**	**2.0**	

3.2.2 LCA methodology

3.2.2.1 Calculation framework

In this study GHG emissions of organic milk production were calculated from cradle-to-farm gate. The production at the analysed farm was assessed for six consecutive years, i.e., one entire crop rotation, using a detailed mass and energy flow model. The calculation guidelines of the IDF were considered noting the differences in the methodology in the following. Emission factors and algorithms from IPCC (2006) and Rösemann et al. (2013) were used for the calculations. It is not part of this study to analyse the uncertainties of these emission factors nor to discuss the use of different emission factors.

3.2.2.2 System boundary, functional unit, and allocation

The system boundary used for the current study includes all agricultural processes in the dairy cow section of the farm in Trenthorst from 2007 to 2012. Upstream processes (production of diesel, lime, silage foil, mineral feed, electricity, liquefied petroleum gas, and transports) and waste management processes (disposal of silage foil) were included. Processing of raw milk and consumption were not considered in the study. Material flows below 5 % of total material flows on unit process level can be cut-off if their environmental contribution is expected to be smaller than 2 % on unit process level (ISO 2006). Consequently, detergents and disinfectants used in the milking parlour were not included in this study.

The product under study was milk produced on the dairy farm. The functional unit used for comparison was 1 kg energy corrected milk (ECM) with an energy content of 3.17 MJ kg^{-1} (IDF 2010). In Germany, the milk year commonly starts in October of the previous year and ends in September, and is combined in the model annually with the previous crop year. That means, for example, that the analysed year 2009 includes the expenses and GHG emissions from the crop production 2008.

During grain production all expenses were allocated to grain, except for processes specifically needed for straw (baling and transportation of straw). As both grain and straw were used entirely within the product system, we assumed that this choice has no effects on overall results.

The allocation between milk and meat was based on energy requirement for live mass gain as given in GfE (2001) based on an approach suggested by Nguyen et al. (2010) in a study on beef production systems. Per kg live mass gain 78.6 MJ gross energy intake are needed. Only feedstuffs required for the live mass increase of the cows and their offspring – and associated processes - were allocated to meat production. The remaining feed - required for lactation, metabolism and movement – was allocated to milk production. This allowed us to include changes in herd size in the allocation. Otherwise changes in herd size from unrelated management choices could be misinterpreted as (in)efficiency. Secondly, since we allocated each age group with different feed demands individually according to their specific daily live mass gain, we considered the impact of offspring on milk production more appropriately. Thus the two co-products of the analysis are milk leaving the farm and live mass increase on herd level, without regard to the disposition of the animals.

Expenses for concentrates for dairy cows were entirely allocated to milk as its amount in the feed ration was solely based on milk yield.

Although infrastructure, i.e., machinery and buildings, contributes significantly to the environmental performance of agricultural production (Nemecek and Erzinger 2005, Koesling et al. 2015) it was not included in the current study. This has been done in many LCA studies on milk production (Haas et al. 2000; Cederberg and Mattson 2000; Cederberg and Stadig 2003; Flysjo et al. 2011; Eide 2002; Thomassen et al. 2008b) and is not necessarily demanded by the IDF guidelines (IDF 2010).

Increases or decreases of soil carbon contents in soils were not accounted for and hence associated emissions and nitrogen flows were not included in this study. These values are of high uncertainty (Gardenas et al. 2011) and site and climate specific. General advice for improvements by crop rotation design and input of organic materials exists (Novak and Fiorelli 2010). Soil carbon sequestration has an important impact on the greenhouse gas balance of agricultural production (Koerber et al. 2009; Petersen et al. 2013). However, we expected that including this factor or machinery and buildings could mask effects of management actions while adding to overall uncertainty.

The impact assessment was performed with the characterization factors for global warming potential for methane of 25 kg CO_2-equivalents (CO_2-eq) and for nitrous oxide of 298 kg CO_2-eq from the IPCC in the 2006 version (IPCC 2006) as noted in the IDF guidelines (IDF 2015).

3.2.3 FARM model and settings for actual calculation

3.2.3.1 Structure

The model FARM (Flow Analysis and Resource Management) has been developed at the Thünen-Institute of Organic Farming to assess material flows and environmental effects of production on the farm level. The model is based on the LCA and material flow analysis software Umberto 5.6© (ifu Hamburg GmbH). The structure of the FARM model is hierarchical. Different perspectives on the life cycle can be assumed, such as an overview of the entire life cycle, focus on material flows between different sections of the farm, or individual work steps in the crop production. Different algorithms for emissions, different settings for energy use for field works and other overarching parameters can be used and compared to assess their sensitivity. Variation of algorithms, data and other settings can easily be conducted by modifying and importing special input files containing these settings. Basic flows and their linkages are shown in Figure 1. Emission substances and algorithms for the calculation of basic flows and emissions used in this study can be found in Table 6.

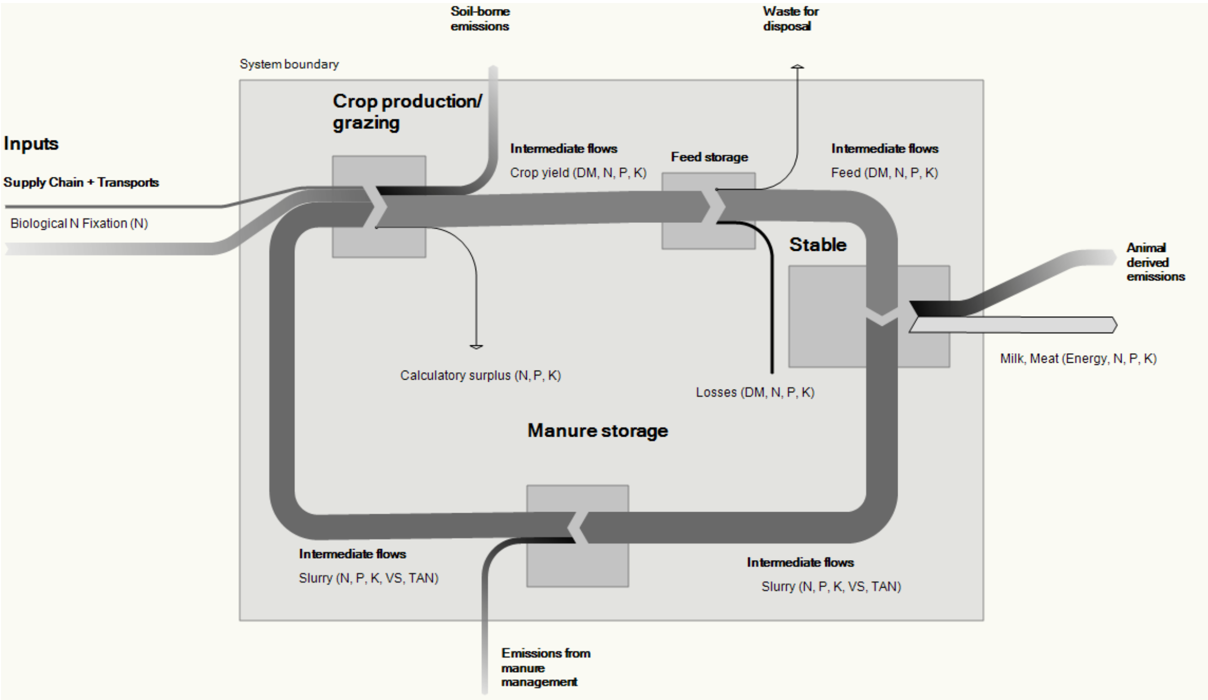

Figure 2: Basic flows and linkages in the FARM model. Rectangles indicate points of even mass balances for the nutrients nitrogen (N), phosphorous (P), and potassium (K). P and K flows are not used to calculate greenhouse gas (GHG) emissions. Emission substances and algorithms for the calculation of GHG emissions are presented in Table 9. DM, VS, and TAN are dry matter, volatile solids, and total ammonia nitrogen, respectively.

Table 9: Emission substances and calculation of basic flows and emissions in the FARM model

Flow	Calculation	Source
Inputs		
Supply Chain		
CO2-eq	$CO_2\text{-eq}_{Input} = kg_{Input} \times CO_2\text{-eq kg}^{-1}$	Althaus et al. (2007)
Transports		
CO_2-eq	$CO_2\text{-eq}_{Transport} = kg_{Input} \times \text{km} \times CO_2\text{-eq ton}^{-1}\text{ km}^{-1}$	Ifu (2005)
Crop production		
Soil-borne emissions		
NH_3	$NH_3\text{-N} = 0.1 \times (\text{Slurry-N} + \text{CR-N} + \text{Seed-N})$	IPCC (2006)
N_2O	$N_2O\text{-N} = 0.01 \times (\text{Slurry-N} + \text{CR-N} + \text{Seed-N} – NH_3\text{-N})$	Adapted IPCC (2006)
Leaching/run-off		
NO_3	$NO_3\text{-N} = 0.3 \times (\text{Slurry-N} + \text{CR-N} + \text{Seed-N} – NH_3\text{-N})$	Adapted IPCC (2006)
Intermediate flows		
Crop yield dry matter	$\text{Crop}_{DM} = \text{Crop component} \times \text{Dry matter content}$	Jeroch (1999)
Feed storage		
Dry matter loss	$\text{Loss}_{DM} = 0.03$ (for concentrates) $\text{Loss}_{DM} = 0.15$ (for roughages)	
Stable		
Animal derived emissions		
CH_4	$CH_4 = GEI \times 0.065$ $GEI = FEED_{GE} \times \text{kg DIDM}^{-1}$	IPCC (2006) IPCC (2006)
NH_3	$NH_3\text{-N} = N_{urine} * 0.197$ $N_{urine} = N_{feed} – N_{milk} – N_{meat} – N_{excr}$ $N_{excr} = DMI \times 0.001 \times (40\ N_{intake} + 6.25^{-1} \times (20 \times DMI + 1.8 \times DMI^2))$	Rösemann et al. (2011) Rösemann et al. (2011)
Intermediate flows		
Slurry VS	$VS = DMI * (1\text{-}XD) * (1\text{-}XA)$	Rösemann et al. (2011)
Slurry TAN	$TAN = N_{urine} * 0.803$	
Manure Storage		
Storage emissions		
CH_4	$CH_4 = VS \times B_0 \times MCF \times 0.67$	IPCC (2007)
N_2O	$N_2O\text{-N} = \text{Slurry-N} \times 0.005$	Rösemann et al. (2011)
NH_3	$NH_3\text{-N} = \text{Slurry-TAN} \times 0.105$	Rösemann et al. (2011)
Grazing		
NH_3	$NH_3\text{-N} = 0.1 \times \text{Droppings-TAN}$	IPCC (2006)
N_2O	$N_2O\text{-N} = 0.02 \times (\text{Droppings-N} – NH_3\text{-N})$	IPCC (2006)
Indirect emissions		
N_2O	$N_2O\text{-N} = 0.01 \times NH_3\text{-N} + 0.0075 \times NO_3\text{-N}$	IPCC (2006)

B_0: Default methane production capacity (0.24 m^3 CH_4 kg VS^{-1})
CR: Crop residues
DM: Dry matter
DIDM: Daily intake dry matter per animal
$FEED_{GE}$: Gross energy content of feed per kg DM (18 MJ for roughages and 19.22 MJ for concentrates)
GEI: Gross energy intake
MCF: Methane conversion factor (10 %)
N: Nitrogen
Nexcr: organic N in feces
TAN: Total ammonia nitrogen
VS: Volatile solids
XA: Ash content of feed (kg kg DM^{-1})
XD: Apparent digestibility of organic matter of feed (kg kg DM^{-1})

Supply chains of intermediate inputs (e.g., silage foil, diesel fuel) are included using datasets from ecoinvent 2.2 (Althaus et al. 2007) within the Umberto software.

Results from the FARM model can be reported both product and area related. Additionally, the environmental performance of each intermediate product, such as crops after harvest or feed after storage, is calculated. This allows not only deeper insight into the production system but also makes the calculation process more transparent.

3.2.3.2 Crop production

Crop production is divided into one module for each crop in the crop rotation. Each module is in turn subdivided into the standard work steps according to KTBL (2014) and can easily be expanded or adapted to local conditions. Diesel consumption can be included both as a total for the entire farm or for each individual work step, depending on the scope of the study and available data. Application of organic fertilizer is specified for each field. Emissions of ammonia, nitrous oxides, and CO_2 from lime use, manure application, and crop residues are calculated with the IPCC (2006) Tier 3 approach (Dämmgen & Hutchings 2008), nitrate leaching based on IPCC (2006) Tier 1.

For this study the diesel demand of crop production, feed storage and feeding was calculated for each work step based on KTBL (2004, 2014) data. The amount of manure and the related nitrogen loads applied to each field were included based on average measured N-contents and applied volumes according to farming records (Table 5).

3.2.3.3 Feed processing

Energy demand for drying (5 MJ kg^{-1} H_2O^{-1}) was taken from the ecoinvent 2.2 database (Nemecek et al., 2007), mixing and milling of the concentrate fractions (0.09 MJ kg^{-1} $grain^{-1}$) was estimated based on expert judgement (personal communication, Boris Martin, Martin GmbH, Bad Lobenstein). Expenses for the ensiling (here only plastic foil) are given as a yearly demand and calculated per kg of silage (3.09 g foil kg^{-1} $silage^{-1}$).

3.2.3.4 Herd structure and products from animal husbandry

With the FARM model each feeding group is simulated separately in terms of group size, ration, and live mass increase on a daily level. Methane emissions from ruminant digestion were calculated separately for each feeding group using the IPCC Tier 2 approach (IPCC 2006, Table 9). Energy for milking (57 kJ kg^{-1} raw $milk^{-1}$) and cooling (104 kJ kg^{-1} raw $milk^{-1}$) is calculated according to LFL (2012).

3.2.3.5 Pasture

Methane emission from ruminant digestion that is related to feed intake on pasture is calculated together with enteric methane emission caused by all other feed intakes. N excretion is calculated separately for stable and pasture based on the share of dry matter intake in the stable and on the pasture. Emissions from urine and dung deposited by grazing animals are calculated using the emission factors in Table 6 and reported aggregated with emissions from manure spreading in the module crop production.

Grass occasionally mowed on pastures is mixed with clover grass from the crop rotation while silage-making. Emissions from work steps (mowing, swathing, transporting) are attributed to this harvest related to dry matter yields and reported aggregated with the module crop production.

3.2.4 Assessment of inter-annual variability

3.2.4.1 Calculation of sample size to gain reliable averages in crop yields

Variation of crop yields leads to a variation in the results. Using average data, the farm's performance can be measured irrespective of inter-annual variation. To calculate the minimum number of years for reliable averages of crop yields, we used the formula for minimal sample size iteratively (Köhler et al. 2012).

$$n = \frac{t\left(1-\frac{\propto}{2}, n-1\right)^2 \times CV^2}{D^2} \qquad (1)$$

n	**Minimal number of years**
t	**Value from t-distribution table for confidence level α and n. N calculated iteratively.**
α	**Risk of type-1-error**
CV	**Coefficient of variation (%)**
D	**Relative margin of error (%)**

The coefficient of variation CV was calculated based on yield data from 2006 till 2013 (Table 4) which covers two additional years after the assessed period to have a more realistic estimate of the crop yield variability. α was set to 0.95. The relative margin of error D was set to 5% meaning that the average yield from n years is within ±5% of the true average and would therefore satisfy a 5% cut-off criterion.

3.2.4.2 Using uncertainty as basis for sensitivity analysis

To obtain more representative results we calculated the model using average activity data, i.e. the arithmetic mean of all parameters from 2 to 6 adjacent years. However, when only a few years are used to calculate the average performance, it is uncertain whether the average is representative in regard to inter-annual variability. The easiest way to quantify this uncertainty is a sensitivity analysis by choosing different values for the average yield and observe the effect on the results. To find sensible upper and lower values for the sensitivity analysis we suggest to permute formula (1) to calculate the uncertainty D from the variability of crop yields:

$$D = \frac{t\left(1-\frac{\propto}{2}, n-1\right) \times CV}{\sqrt{n}} \qquad (2)$$

The t-value at a given α-value is only depending on the number of years being 12.7, 4.3, 3.2, 2.8, 2.6 for 2, 3, 4, 5, and 6 years, respectively. For low n and high variability, the resulting D can be > 100%. We suggest calculating upper and lower yields by formula (3) and (4), respectively.

$$Upperyield = averageyield \times (D/100 + 1) \quad \textbf{(3)}$$

$$Loweryield = averageyield \times (D/100 + 1)^{-1} \quad \textbf{(4)}$$

To illustrate the procedure, for clover yield based on 2 years and a CV of 12.6 % the uncertainty D is 113 %. The average yield for the years 2006 and 2007 was 7.15 t DM ha^{-1}. The upper and lower values for the sensitivity analysis are 15.23 t DM ha^{-1} and 3.36 t DM ha^{-1}, respectively. For six years D is 13 %, the average yield for the years 2006 till 2011 is 7.2 t DM ha^{-1}, the upper and lower values for the sensitivity analysis are 8.14 t DM ha^{-1} and 6.37 t DM ha^{-1}, respectively.

We used the upper and lower values of the average yield of each crop for every combination from two to six consecutive years (Table 10) for the sensitivity analysis and calculated the GHG emissions per kg ECM.

Table 10: Relative uncertainty of average yields for different number of assessed years based on coefficient of variation (CV) from yields in Trenthorst 2006-2013

Crop	CV	Uncertainty of average yield based on the number of years				
		2	3	4	5	6
Clover-grass	12.6 %	113,2%	31,3%	20,1%	15,7%	13,2%
Silage maize	37.2 %	334,2%	92,4%	59,2%	46,2%	39,0%
Winter wheat	23.5 %	211,1%	58,4%	37,4%	29,2%	24,7%
Triticale	25.0 %	224,6%	62,1%	39,8%	31,0%	26,2%
Oats/field beans	21.3 %	191,4%	52,9%	33,9%	26,5%	22,4%

3.3 Results

3.3.1 GHG emissions from milk production

The calculated GHG emissions for the production of 1 kg ECM showed variation over the years. Results varied between 0.88 kg CO_2-eq kg^{-1} ECM^{-1} in 2009 and 1.09 kg CO_2-eq kg^{-1} ECM^{-1} in 2011 (Figure 2). The mean was 0.97 kg CO_2-eq kg^{-1} ECM^{-1} and the standard deviation of the mean (SDM) was ±0.07 kg CO_2-eq kg^{-1} ECM^{-1} and the coefficient of variation (CV) was 7.5 %.

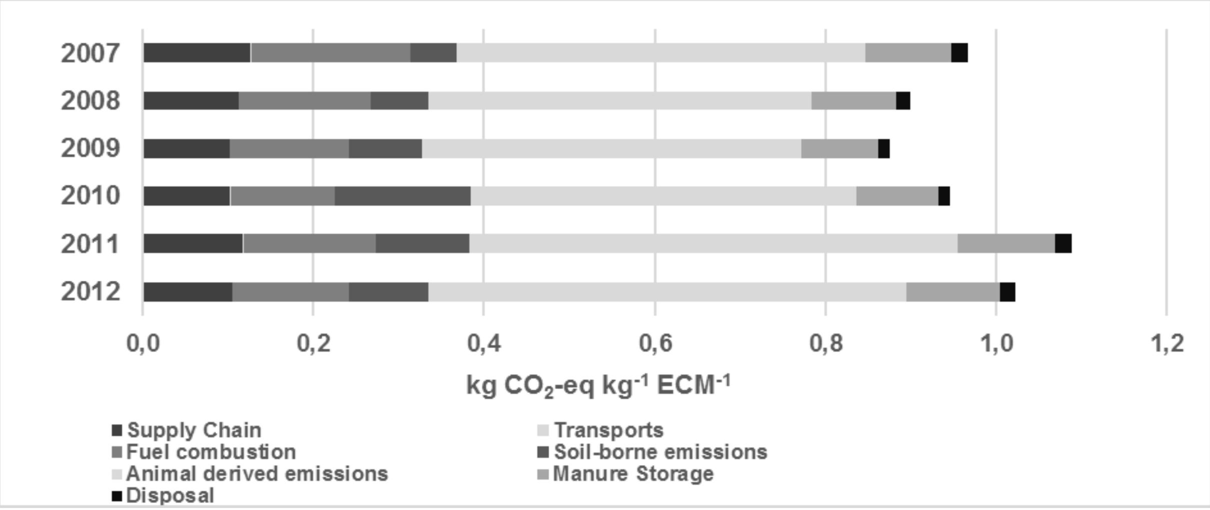

Figure 3: Contribution of each life cycle phase to global warming potential per kg energy corrected milk (ECM) for the years 2007 till 2012 at the research station of the Thünen Institute of Organic Farming in Trenthorst.

3.3.2 Contribution of emission sources

The absolute contribution of animal derived emissions (methane from ruminant digestion and indirect N_2O from ammonia emissions in the stable) to overall results was 0.49±0.05 kg CO_2-eq kg^{-1} ECM^{-1} ranging from 0.45 to 0.57 in 2008 and 2011, respectively (Figure 2). Fuel combustion contributed 0.15±0.02 kg CO_2-eq kg^{-1} ECM^{-1} with a range from 0.12 to 0.19 in 2010 and 2007. The supply chain of intermediate products contributed 0.11±0.01 kg CO_2-eq kg^{-1} ECM^{-1} (range: 0.10 to 0.13 in 2009 and 2007). Emissions from manure storage contributed 0.10±0.01 kg CO_2-eq kg^{-1} ECM^{-1} (range: 0.09 to 0.12 in 2009 and 2011). Soil-borne emissions from fertilization and crop residues contributed 0.10±0.04 kg CO_2-eq kg^{-1} ECM^{-1} with the range from 0.05 to 0.16 in 2007 and 2010. The greenhouse gas emissions from disposal were 0.02±0.002 kg CO_2-eq kg^{-1} ECM^{-1} and from transports 0.0008±0.0002 kg CO_2-eq kg^{-1} ECM^{-1}.

The highest average contribution to GHG emissions per kg ECM came from methane emissions from ruminant digestion of the animals (50.9±2.3) %, followed by emissions from fuel combustion (15.5±2.3) %, manure storage (10.5±0.3) %, supply chain of intermediate products (production of diesel fuel, silage foil, lime fertilizer, and electricity) (11.5±1.0) % and direct emissions from soil fertilization (9.8±3.5) %. Emissions from disposal (1.8±0.2) % and transports were less important in relation to the overall results.

When partitioning the calculated impacts between the feeding groups, cows (lactating + dry cows) had the highest mean contribution to overall results with 0.88±0.07 kg CO_2-eq kg^{-1} ECM^{-1} with a range between 0.79 and 1.00 in 2009 and 2011, respectively (Figure 4). Young stock and heifers combined contributed 0.056±0.023 kg CO_2-eq kg^{-1} ECM^{-1} with a range between 0.023 and 0.083 in 2007 and 2012, respectively. Suckling calves and calves together contributed 0.028±0.01 kg CO_2-eq kg^{-1} ECM^{-1} on average.

Over all years, lactating cows were responsible for 83.9±1.5 % of all milk related GHG emissions of the analysed farm. Dry cows contributed about 7.4±0.5 %. Young stock and heifers combined contributed 5.7±2.2 % and calves aged 0 days to 180 days contribute 3.0±0.9 %.

The contribution of the feeding groups has to be seen in context of the allocation performed. As most live mass increase is done by the calves, young stock, and heifers, the allocation between expenses for milk (here maintenance) and live mass increase is higher compared to the cows. On average, the expenses allocated to live mass increase are 9 % for cows, 63 % for young stock and heifers, and 95 % for calves and suckling calves. For the entire farm, 69 % of emissions are allocated to milk and 31 % are allocated to live mass increase.

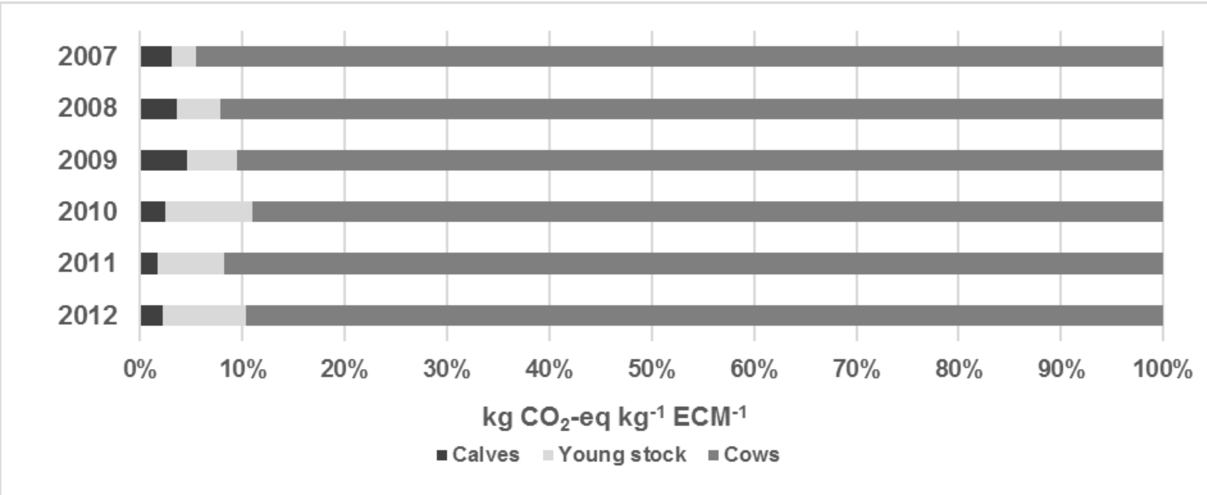

Figure 4: Contribution of the different feeding groups to greenhouse gas emissions per kg energy corrected milk (ECM) for the years 2007 till 2012 at the research station of the Thünen Institute of Organic Farming in Trenthorst.

3.3.3 GHG emissions from intermediate products

When the feed components are compared at the point of entering the stable, the production of concentrate feed had the highest emissions with 0.36±0.08 kg CO_2-eq per kg dry matter feed produced (kg^{-1} DM^{-1}) ranging between 0.25 and 0.45 in 2011 and 2008, respectively (Figure 4). Maize silage had GHG emissions of 0.20±0.08 kg CO_2-eq kg^{-1} DM^{-1} ranging between 0.12 and 0.33 in 2007 and 2010, respectively. The production of grass silage lead to GHG emissions of 0.28±0.03 kg CO_2-eq kg^{-1} DM^{-1} ranging between 0.24 and 0.31 in 2007 and 2009, respectively. The CV of GHG emissions from feedstuff production on the farm was highest for maize silage with 39 %, followed by concentrate (22 %), and grass silage (10 %).

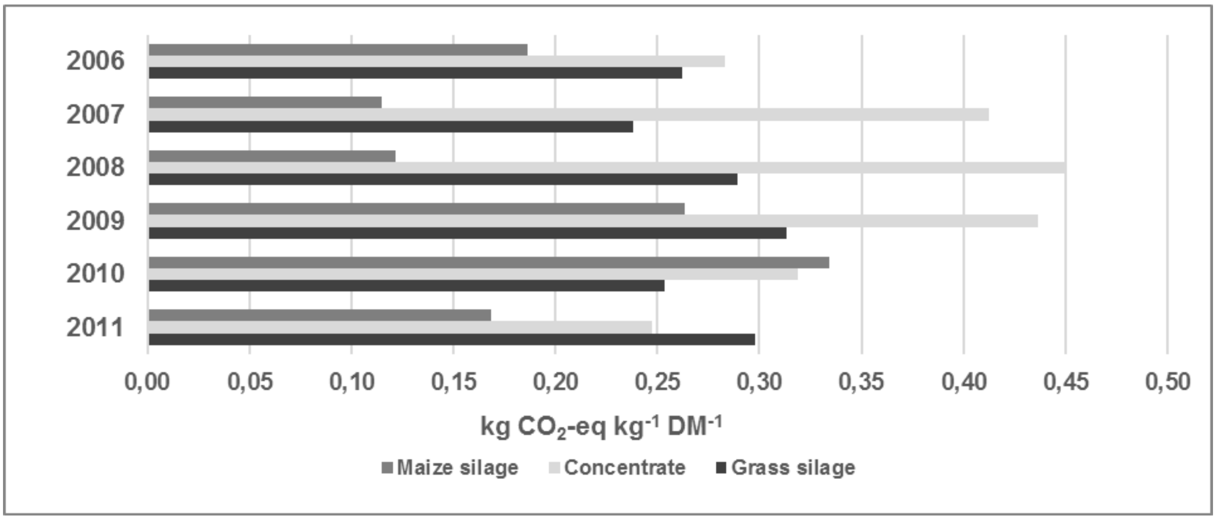

Figure 5: Global warming potential per kg dry matter of feedstuffs from 2007 till 2012 at the research station of the Thünen Institute of Organic Farming in Trenthorst. Averages were 0.20, 0.36, and 0.28 kg CO2-eq kg^{-1} DM^{-1} for maize silage, concentrates, and grass silage, respectively.

3.3.4 Number of years needed to assess due to variation of crop yields

The sample size needed to gain reliable averages of crop yields due to the variation based on crop yields between 2006 and 2013 are calculated from Formula 1. Clover/grass mixture had the lowest sample size with n = 27, meaning that an average derived from 27 years of clover/grass production would lie within ±5 % of the real average of the production system. The production of grains varied as such oat/field bean mixture would require a sample size of n = 72, winter wheat of n = 87, and triticale of n = 99. Maize production had the highest variation and the resulting sample size is n = 215.

3.3.5 Uncertainty as basis for sensitivity analysis

The results based on average data are presented in Figure 6. The effect of upper and lower values for average yields are indicated as black diamonds. The average difference between upper and lower result is 0.24 kg CO_2-eq kg^{-1} ECM^{-1} (27 %) for two years' averages and gradually decreases to 0.03 kg CO_2-eq kg^{-1} ECM^{-1} (3.6 %) for the six years' average.

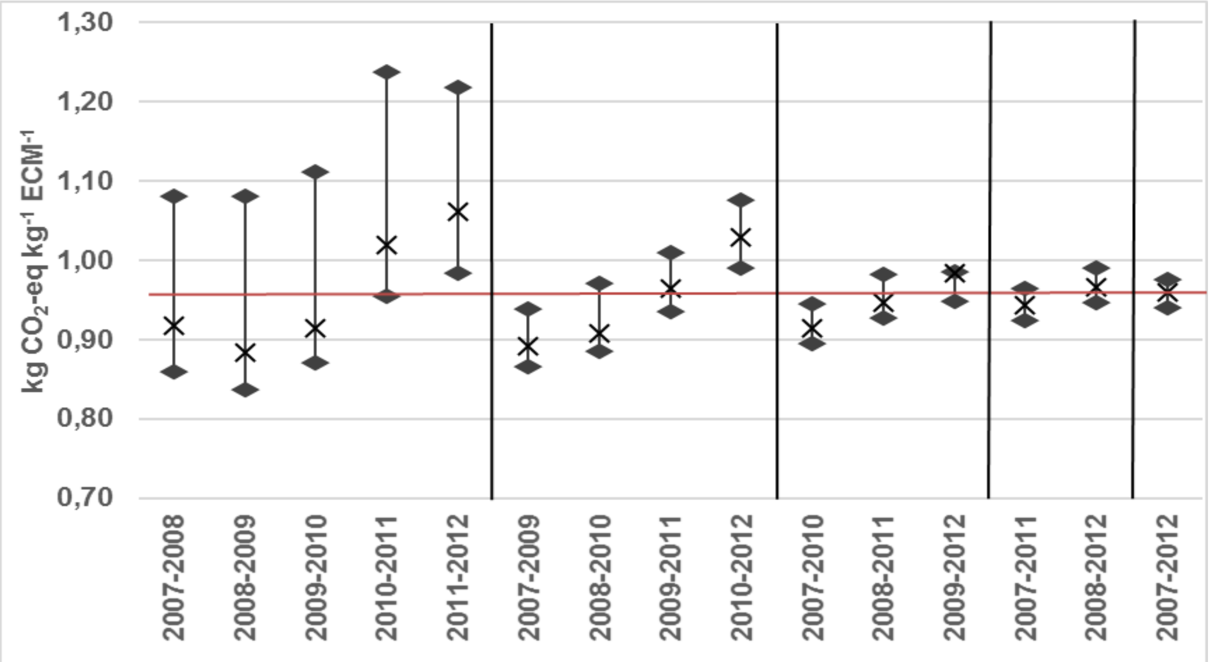

Figure 6: Bandwidth of global warming potential per kg energy corrected milk for the years 2007 till 2012 using simulated crop yields (black diamonds) based on average yields (x) for different time spans (2-6 years) at the research station of the Institute of Organic Farming in Trenthorst.

3.4 Discussion

3.4.1 Impacts from milk production

Production data, i.e., crop and milk yield level, on the research farm varied considerably (Table 4, 5, and 6); therefore, the inter-annual variation of the product related GHG emissions of milk appears plausible. Since to our knowledge no study exists on inter-annual variation of product related GHG emissions for dairy farming the variation of the overall results cannot be compared to other studies. In comparisons of different farms, Guerci et al. (2013) found a range of 0.55-1.91 kg CO_2-eq kg^{-1} ECM^{-1} in Denmark, Germany and Italy (n=12). Frank et al. (2013) found a bandwidth of 0.84-1.4 kg CO_2-eq kg^{-1} ECM^{-1} in 12 organic dairy farms in South- and West-Germany and a range of 0.93-1.25 kg CO_2-eq kg^{-1} ECM^{-1} in 12 conventional farms in the same regions. Our results are within the ranges found in these studies.

Based on the high detail of data collection we expected to understand the sources of variability. However, the number of years is not enough to find statistical relationships between the production data on farm and total product-related GHG emissions. For example, milk year 2012 had a good milk yield with relatively low numbers of offspring. The relevant crop year 2011 had a good harvest combined with relatively low N inputs. Therefore, we expected that milk produced in 2012 would have lower than average product-related GHG emissions. Instead, 2012 has the second highest product-related GHG emissions in the analysed period.

We conclude that the calculated high dry matter intake of lactating and dry cows lead to higher GHG emissions. In the milk year 2010, 89 % of the milk was produced with only 82 % of daily dry matter intake of lactating cows, compared to 2012. The reasons may be that our analysis did not take into account variation of feed quality as this data does not exist consistently throughout the period, and feed intake may vary due to changes in feed quality (Allen 2000).

3.4.2 Contribution analysis

Methane emissions from ruminant digestion contribute 50 % to overall GHG emissions. Since the system under study is a low external input organic farm this result is plausible and comparable to other studies of GHG from organic milk production (Cederberg & Mattson, 2000, Frank et al. 2013). The amount of methane produced per kg milk can be decreased by increasing the milk production per cow (Brade et al. 2008), if changes in inputs are moderate, e.g., if milk yield is increased by improved health management or improved roughage quality (Frank et al. 2015, Paulsen et al. 2015, Warnecke et al. 2014). However, whether, e.g., a higher demand in concentrate feed leads to a shift of emissions from ruminant digestion to feed production, depends on milk yield gains, yield levels in feedstuff production and farm management, so that specific scenarios must be calculated to judge on the right intensity to reach improved environmental goals.

Emissions directly related to the animals, such as methane emissions from ruminant digestion and manure storage (via the amount of manure excreted), show less relative variation over the years with 4.4 % and 2.7 %, respectively, than emissions related to crop production such as soil-borne emissions (36 %) and fuel combustion (15 %). This is likely due to the fact that in most calculations feed intake and milk and manure output are in direct relationship to milk yield (Gruber et al. 2006), whereas crop yields are not only influenced by management decisions but also by external factors such as weather conditions. Furthermore, low yields can be compensated by changes in herd size, e.g., reduction of young stock, without affecting milk output of the farm.

An important reason for the varying contribution of the different feeding groups to the overall results lies in the animal numbers of the feeding groups. The dominating contribution of cows to overall results is not surprising considering their number and the related amount of feed intake in comparison to the calves, young stock and heifers (Figure 4). As a result of the relatively constant feed supply, the relative variation of the cow's contribution is low compared to other feeding groups.

The contribution of suckling calves and calves may seem high in relation to young stock and heifers as not only their dry matter intake is lower, but also their head count is lower. The reason for the high contribution is the feeding of raw milk to the suckling calves. They consume between 4 % and 7 % of the total produced milk on the analysed farm. Therefore, all emissions associated with the production of milk for calves are allocated to the calves.

As described in Section 2.2.2, the FARM model calculates allocation between expenses for milk and expenses for meat separately for each feeding group. The percentage of energy needed for live mass increase from total energy demand is higher for suckling calves and calves compared to young stock, heifers, and cows. This means that the major share of milk consumed by suckling calves is actually allocated to meat production.

The allocation between milk and meat, we have used in our study, is in general not so different from the procedure outlined in IDF (2010). Simply using live mass gains and milk yields from Table 6 in the IDF procedure yields an average allocation factor of 62 % of emissions to milk and 38 % to meat over the six years. However, the approach from the IDF does not take into account stock changes between the years as it only looks at sold animals, and secondly assumes the same ration across all feeding groups as a partition of feeding groups does not take place. However, especially concentrates are mainly used for lactating cows while the feed for followers is more extensive. This can be taken into account with our approach. The relative high values for allocation towards meat can be explained by having Red Holstein Double Usage animals, relative low milk yields, and more followers kept in the farm than needed for replacement.

3.4.3 Variability in animal husbandry

Variation in milk yield influences the GHG balance of milk in two aspects. First, the scaling of all emissions to the functional unit is performed based on overall sold milk in this study. More milk leaving the farm at a given emission level leads to lower emissions per kg ECM.

Secondly, changes in feed quality and digestibility influence dry matter intake and milk yield as well as methane emissions from ruminant digestion (Warnecke et al. 2014). Multiple studies exist on the adequate use of algorithms for CH_4 prediction (Ellis et al. 2010, Schulz et al. 2013), but for the purpose of this study - to analyse effects of crop yield variability - feed quality was not included. The used algorithm for the prediction of ruminant methane emissions does not include feed quality parameters and digestibility, but instead assumes a pure linear relationship between gross energy intake and methane emission (Table 9) and includes relative uncertainty of 15 % to either side (IPCC 2006). But the effects of the given uncertainty of algorithms and of varying feed quality are not part of this study.

3.4.4 Variability within crop production

The high variability of crop yields led to extreme values of the calculated sample size to obtain reliable averages of crop yields for the current farm situation. This is neither practicable nor sensible. Advancements in breeding of plants and animals, and changes in management practices and infrastructure over such a period make the description of consistent systems impossible. While the variability in this study seems high, FAO statistical data also suggests that this is typical for agricultural systems. The yields per hectare for maize during 2003 and 2013 – averaged for entire Germany – had a CV of 9.4 % (FAOSTAT 2015) which would lead to a minimum of 16 years to be calculated. In organic farming and extensive plant production – as visible also in the analysed farm – variation of yields might be generally higher compared to conventional plant production due to restricted possibilities of fertilization, weed and pest control.

Another source for the variability of GHG emissions from feed production is the amount of applied slurry and farm yard manure, which also depends on changes of the herd size. For example, in the milk year 2009 the number of cows increased by 10 % compared to the previous year, leading to a higher slurry production in that year, and therefore to higher slurry application in the following crop year. With yields lower than average in crop year 2010, dry matter related GHG emissions from maize silage production in 2010 were higher than average (Figure 5).

Generally, GHG emissions of feed production related to dry matter vary much more than GHG emissions per kg ECM. This can be explained by the adaptability of feeding management to changing resources. In the analysed farm the substitution of maize with grass silage in milk year 2011 (Table 8) due to low maize yields in crop year 2010 (Table 4) can serve as an example. Potential energy deficits were compensated with concentrate feed. These aspects of variability and compensation make farm comparisons difficult.

In concluding, if different farming management practices and their effects on GHG emissions of production are to be compared, the representativeness of the assessed time period is crucial to the interpretation of the results. While emissions from a specific year maybe estimated correctly to be lower in one system compared to another system, the result does not necessarily reflect the superiority or preferability of the system.

3.4.5 Assessment of inter-annual variability

3.4.5.1 Calculation of sample size based on own yield data

The assumption that the CV from 8 years represents the population's variation might be criticised. Sensible alternatives are using crop yield variability from statistical databases (e.g. on the regional or national level) or long-term experiments. However, this would result in a further lack of representativeness in regard to local conditions (weather, soil, crop rotation) and management (pest and fertilizer management). We think that the use of the actual yields and their observed variation provide the best picture of the assessed farm.

3.4.5.2 Using uncertainty as basis for sensitivity analysis

Differences between upper and lower values in the sensitivity analysis decrease with an increase in assessed years, which can be expected as the uncertainty gradually decreases. Depending on the scope of the study, one can choose which level of uncertainty is acceptable. However, we think that an uncertainty of over 25 % just from the variability of crop yields is not sufficiently precise as a result and therefore, based on our results, at least 4 or 5 years should be analysed when assessing the environmental performance of agricultural systems.

We assume that overlapping bandwidths mean that the true environmental performance of the studied farm is included in the temporal scope, and is therefore representative with regard to inter-annual variation of crop yields. It is thus suitable for use for comparisons between different real farms. Since crop yields have a high variation on the studied farm, only timespans of four years or more show overlapping bandwidths of all results. We expect that fewer years could suffice for the calculation of representative results under conditions with smaller yield variation. However, the uncertainty of the results would remain very high unless the CV of the crop yields would be estimated much lower.

This approach can and should be extended to other sources of variability such as milk yield and diet composition. However, the relationship between scaling to the functional unit milk and variability of milk production on herd level and their effect on GHG emissions should be assessed beforehand. Changes in diet composition must also be addressed by the appropriate use of methane prediction algorithms (Piatkowski et al. 2010).

In a practical application, different farms or farming systems have different environmental impacts when the bandwidths of their results do not overlap more than

5%, which can be justified by our cut-off rule. Otherwise, differences between the farms or farming systems cannot be regarded as significant. Significant differences between farms can only be found when analysing two systems with large differences, or analysing sufficient years to minimize the uncertainty of results. If the goal of the analysis is to differentiate between different farming systems or management practices, we believe it to be possible to analyse multiple farms instead of multiple years. However, since different farming systems may react differently to weather conditions, it may still be necessary to assess the temporal representativeness.

3.5 Conclusion

The dairy farm of the Thünen-Institute of Organic Farming is used as an example for organic milk production in northern Germany. GHG emissions for six consecutive years were assessed to analyse the inter-annual variation of production data from a product-related perspective. A variation of 7.5 % of overall GHG emissions per kg ECM confirms that the analysis of only one production year is insufficient for farm comparisons or of comparisons of farming systems or management practices.

It has been shown that variability of crop yields influences management practices concerning diet composition or herd structure, while management practices concerning herd structure or diet composition may influence crop yields. Quantifying uncertainty based on the number of years included and the variability of crop yields can help with interpreting the average environmental performance of a farm and allow for results that can be used for farm comparisons.

In order to ensure reliable results, temporal representativeness should be explicitly dealt with in guidelines for the assessment of environmental performance of agricultural production. In further research, this approach should be extended to other sources of variability, e.g., milk yield of dairy cows, diet composition, and be tested under practical conditions, e.g., assessment of a group of practical farms over multiple years.

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4 Discrimination of milk carbon footprints from different dairy farms when using IPCC Tier 1 methodology for calculation of GHG emissions from managed soils

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Abstract

Quantification of environmental performance of dairy farms should allow comparisons between farms. We assess whether IPCC Tier 1 methodology for emissions from soil management is sufficiently precise to analyse and differentiate the carbon footprint of milk production between practical dairy farms, and whether we can correctly identify which farms have the lowest and the highest GHG emissions per product unit, respectively.

We used data from 20 Norwegian dairy farms which are very similar in structure but differ in organic/non-organic management and the share of peat soil of their farmland. We assessed the uncertainty of the carbon footprint by running Monte Carlo simulations with uncertainty ranges given in Tier 1 of the IPCC guidelines. The carbon footprint is considered different when 95 % of all Monte Carlo iterations assume one farm to have higher product related GHG emissions than the farm in comparison.

The uncertainty of results in the single farms, expressed as two-times standard deviation divided by the median result, ranges between 4.2 % and 15.3 %. This means that 95 % of values in the resulting distribution of one farm are within a range of 4-16 %

of the median of that farm. Farms can be discriminated when the variation of the carbon footprint is higher than the uncertainty of farm related emissions. From all 190 direct comparisons of two farms in the study, 78 % are significantly different.

For this uncertainty assessment, it must be established that background processes, especially the datasets for import feed, can be judged covariant in order to prohibit them to influence the comparison between farms. Secondly, the uncertainty ranges used for the calculation must be appropriate for the assessed systems.

We were able to accept the hypothesis, that a significant differentiation of the milk carbon footprint between farms is possible with an IPCC Tier 1 approach, for a majority of our comparisons and found a difference of above 8.7 % sufficient to establish significance.

Keywords: sensitivity, uncertainty, IDF, CF, variability, Monte Carlo

Highlights:

- Variation between farms is higher than uncertainty of GHG emissions from managed soils in 20 Norwegian dairy farms
- IPCC Tier 1 methodology is suitable for the carbon footprint of dairy products
- Covariance of emissions in the supply chain crucial for the discrimination between farms of different production systems
- We can identify farms with lowest and highest product-related GHG emissions from a set of 20 Norwegian dairy farms

4.1 Introduction

Environmental performance plays an increasing role in agricultural production. Livestock production has intensive land demands, shows N and P overflow in intensive regions, and ammonia and greenhouse gas emissions during animal husbandry and production of feedstuff. Carbon footprint of milk production is in special focus due to high methane emissions of ruminants, emissions from grazed pastures, forage production on farm, emissions from open stables and variable manure storage and management (Warnecke et al. 2014, Bornesmo et al. 2013, Kristensen et al. 2011, Novak and Fiorelli 2010, Gerber et al. 2010, Amon et al. 2006). With the quantification of the carbon footprint of their products based on whole farm level approaches, farmers could be incentivised to mitigate their emissions. Also, political options to foster climate friendly production might be developed by the results, when based on widely accepted methodology (Pirlo 2012).

With the International Dairy Federation (IDF) guidelines for carbon footprints, an accessible framework for the calculation of the greenhouse gas (GHG) emissions associated with dairy production exists (IDF 2015). Examples of the world-wide use of these guidelines can be found in Dalgaard et al. (2014), Daneshi et al. (2014), Gollnow et al. (2014), and Jayasundara and Wagner-Riddle (2014). The IDF guidelines reference the Tier structure from the IPCC guidelines, where Tiers 1, 2, and 3 use increasingly detailed methods for the calculation of direct emissions (IPCC 2006, Chapter 1). According to the IDF guidelines "[f]or the purpose of achieving consistency in dairy LCAs, it is agreed that at least a Tier 2 approach is necessary" (IDF 2015, p. 26).

The IPCC Tier structure in the dairy sector applies firstly to direct emissions from livestock, which include methane from ruminant digestion and both direct and indirect GHG emissions from manure handling and storage. Secondly, IPCC applies to emissions from managed soils, which include direct nitrous oxide (N_2O) emissions, indirect N_2O emissions due to ammonia emissions (NH_3) and nitrate leaching, and direct carbon dioxide (CO_2) emissions from lime. Additionally, CO_2 emissions from managed grassland for both mineral and organic soils are part of the IPCC methodology.

N_2O emissions from managed soils in the supply chain due to imports of feed are not part of the IPCC guidelines as their purpose is the use in national GHG inventories (National Inventory Reports). From an LCA practitioners' viewpoint, these emissions must be included in the analysis of a carbon footprint as well as other GHG emissions in the supply chain. The IDF guidelines fail to provide clarity whether IPCC Tier 2 should also be followed for the calculation of GHG emissions during the production of import feed.

For methane from ruminant digestion a simple to use Tier 2 method is available from the IPCC guidelines (IPCC 2006, Ch. 10). It considers the gross energy intake of the animals and assumes uncertainty to either side of ~15 %. Contrastingly, for calculation of N_2O from managed soils suitability of Tier 2 methodologies are under discussion, e.g. the National Inventory Report in Germany still lacks methodology to use higher than Tier 1 for this emission source (Haenel et al. 2016). Peter et al. (2016) argue that at regional and sub-regional levels, Tier 1 would not "always [be] sufficiently accurate to account for spatial variability of GHG emissions due to different soil, climate, and management practices". They suggest using a "medium-effort" Tier 2 model, described in Bouwman et al. (2002), for calculation of emissions in crop production. This model could satisfy demands set up by IDF (2015).

Despite the demand for IPCC Tier 2 methodolgy, many carbon footprint models for agriculture still rely on using Tier 1 methodology for the estimation of N_2O emissions from managed soils (Katsch, Osterburg 2016). Adding to that, data collection is already very demanding for whole-farm models assessing GHG emissions in dairy farming. An even higher data demand for using Tier 2 methodology would make the use of whole-farm models more unlikely and could therefore fail to promote GHG assessments as widely accepted tool. Consequently, it is of interest to know whether Tier 1 methodology developed for national GHG inventories are sufficient on farm level and provide results that are transparent, reproducible and fair, especially in the context of a comparative assessment of product related GHG emissions.

In several studies, Monte Carlo simulations have been used to assess the effects of uncertainty of emission factors on the calculation results. The focus of these studies was on finding key variables that induce uncertainty (Basset-Mens et al. 2009, Ross et al. 2014), uncertainty from choices of methods (Zehetmeier et al. 2014), or solely the effect of emission factor uncertainty (Chen & Corson 2014). To our current knowledge no study exists, that analyses the effects of uncertainty on the comparability of results between farms – even though this comparability is one of the key aims of carbon footprints (ISO 2006, IDF 2015).

Aim of the study: We want to assess whether IPCC Tier 1 methodology for direct and indirect N_2O emissions and for CO_2 emissions from soils is sufficiently precise to analyse and differentiate the carbon footprint of milk production between practical dairy farms. We want to correctly identify which farms have the lowest and the highest GHG emissions per product unit, respectively. For our comparison we will use data from 20 Norwegian dairy farms which are very similar in structure but differ in organic/non-organic management and the share of peat soil of their farmland.

The hypothesis for this study is that a significant differentiation of farms is possible with a Tier 1 approach for direct and indirect N_2O emissions and for CO_2 emissions from soil. Methodologically, this is the case when the variation of the carbon footprint of milk between farms is higher than the uncertainty of results. The uncertainty is derived from Monte Carlo simulations with uncertainty ranges given by IPCC Tier 1 methodology as input. Based on this data, we accept our hypothesis when 95 % of all Monte Carlo iterations assume one farm to have higher product related GHG emissions than the farm in comparison.

4.2 Material and Methods

4.2.1 Data generation and description of the dairy farms

In the project Enviromilk, GHG emissions, energy and nutrient flows in three consecutive years on 20 Norwegian dairy farms were determined. 10 organic (certified according to Norwegian National Regulations (2017) for organic farming which are based on EU Regulation 834/2007) and 10 non-organic[2] dairy farms were compared. They were typical farms on the Norwegian west coast (Møre og Romsdal County). Farm data were derived by official databases, farm accounts, farm records and interviews with the farmers. The selected farms in both groups differed in cattle numbers, milking yield, farm area, fertilization intensity and share of concentrates. 50 % of the organic farms and 40 % of the non-organic farms had some areas with peat soils (Table 11). In our study, peat soil is defined as soil with at least 40 % content of organic matter in the top soil layer with at least 20 cm thickness (Sveistrup 1984).

Most of the roughage was produced on the farms with some import in years with unfavourable yield conditions. Concentrates were completely imported on all farms. On cultivated area, only grass and grass-clover leys are grown. In some farms, cereals were used as cover-crop when establishing new meadow and harvested as silage. The grazing period was usually not more than three months for dairy cows and four for heifers in summer time. Dairy cows grazed either on fully cultivated land, in fenced uncultivated pastures, or in forest or mountain area outside the farmland. These environments, as well as their different management and the grazing activities within, were covered by detailed data collection. In the indoor season, the cattle are mainly fed forage and concentrates. A detailed description of the farms, the regional structure of dairy farming and methods of data acquisition can be found in Koesling et al. (2015) and Koesling (2017).

Table 11: Average data and ranges of the analysed farms in the Enviromilk project.

Parameters	Units	Non-organic	Range	Organic	Range
Farms	n	10	-	10	-
Weighted farm area[a]	ha	33.0	17.6-85.1	37.7	14.3-89.2
Off-farm area[b]	ha	28.20	0.7-64.8	24,9	5.9-63.4
Dairy system area	ha	59.3	33-150	61.4	20-154
Farms with peat soil	n	4	-	5	-
Average peat area on farms with peat soil	ha (share)	12 (29 %)	6-24 (22-42 %)	11 (22 %)	3-32 (4-45%)

2 The term „conventional agriculture” is often used to refer to non-organic agriculture. However, the term “conventional” is not defined. “Non-organic” in this text refers to all types of agriculture that are not explicitly organic according to EU Regulation 834/2007.

Milking cows	n	29.5	13.8-67.6	29.4	15.3-65.5
Milk delivered per cow[c]	kg ECM[d] cow^{-1}	7301	6408-8223	5490	2751-7317

[a] Weighted farm area = fully cultivated land + 0.6 x surface cultivated land + 0.3 x native grassland. See also Koesling (2017).

[b] Area used for production of imported feed.

[c] Milk delivered includes sold milk and private use

[d] ECM: Energy-corrected milk (Sjaunja et al. 1991)

4.2.2 Modelling and Emission factors

The calculation of the carbon footprint of milk production was performed in a cradle-to-farm gate life cycle assessment approach with the model FARM (Flow Analysis and Resource Management, Schueler & Paulsen 2012, Schueler et al. 2017) in the umberto software (ifu Hamburg GmbH). The model has been adapted to reflect Norwegian conditions. The basic flows and algorithms are shown in Table 12.

Table 12: Basic flows in the FARM model in the Norwegian version (FARMnor) and their calculations.

Flow	Calculation	Source
Crop production		
Soil-borne emissions		
NH_3	$NH_3\text{-}N = 0.1 \times (\text{Crop Residue-N} + \text{Seed-N}) + NH_3\text{-}N_{Slurry}$	IPCC (2006)
$NH_3\text{-}N_{Slurry}$	$NH_3\text{-}N_{Slurry}$ = 0.4 x TAN spread (spreading plate) + 0.36 x TAN spread (spreading plate with extra water) + 0.3 x TAN spread (trailing shoe)	Rodhe & Karlsson (2002)
N_2O	$N_2O\text{-}N = EF1^A \times$ Mineral Fertilizer-N	IPCC (2006)
	$N_2O\text{-}N = EF2^A \times$ Slurry-N	IPCC (2006)
	$N_2O\text{-}N = EF3^A \times$ Crop residue-N $N_2O\text{-}N = EF4^A \times$ N from droppings during grazing $N_2O\text{-}N = EF5^A \times$ peat soil area	IPCC (2006) IPCC (2006) IPCC (2006)
CO_2	$CO_2\text{-}C = EF6^A \times$ Farm area on peat	IPCC (2006)
Crop residue-N	Crop Residue-N = Above Ground N + Below Ground N Above Ground N = $AGDM^A \times AGDM\text{-}N^A$ Below Ground N = $BGDM^A \times BGDM\text{-}N^A$	IPCC (2006)
Feed storage		
Dry matter loss	$Loss_{DM}$ = 0.03 (for concentrates) $Loss_{DM}$ = 0.15 (for roughages)	Estimation
Stable		Estimation
Animal derived emissions		
CH4	$CH_4 = (1.28 \times \text{kg DIDM} - 1.47) \times D^B$ $CH_4 = (3.74 \times \text{kg DINDF} - 2.76) \times D^B$	Storlien et al. (2014) Storlien et al. (2014)
NH3	$NH_3\text{-}N = N_{urine} * 0.197$ $N_{urine} = N_{feed} - N_{milk} - N_{meat} - N_{excr}$ $N_{excr} = DMI \times 0.001 \times (40\ N_{intake} + 6.25^{-1} \times (20 \times DMI + 1.8 \times DMI^2))$	Rösemann et al. (2011) Rösemann et al. (2011)
Intermediate flows		
Slurry VS	$VS = DMI * (1\text{-}XD) \times (1\text{-}XA)$	Rösemann et al. (2011)
Slurry TAN	$TAN = N_{urine} \times 0.803$	Rösemann et al. (2011)
Manure Storage		
Storage emissions		

CH_4	$CH_4 = VS \times B_0 \times MCF \times 0.67$	IPCC (2006)
N_2O	$N_2O\text{-}N = \text{Slurry-N} \times 0.005$	Rösemann et al. (2011)
NH_3	$NH_3\text{-}N = \text{Slurry-TAN} \times 0.105$	Rösemann et al. (2011)

A	see Table 3 for definition and values
B	Each formula is used in half of the calculations
AGDM	Above Ground Dry Matter
BGDM	Below Ground Dry Matter
B_0	Default methane production capacity (0.24 m^3 CH_4 kg VS^{-1})
D	Average number of animals x 365
DM	Dry matter
DIDM	Daily intake dry matter (DM)
DINDF	Daily intake non detergent fibre (NDF)
MCF	Methane conversion factor (0.1)
N	Nitrogen
TAN	Total ammonia nitrogen
VS	Volatile solids

The FARM model is a hierarchically structured flow model. Inventory flows and emissions from external inputs to the farm as import feed, diesel fuel, silage foil, electricity, and fertilizer are approximated using the ecoinvent life cycle inventory (LCI) database (Ecoinvent 3.2 2015). The dataset for barley was adapted to reflect Norwegian cultivation practise and yields (Bonesmo 2012). Material flows within the farms were checked for plausibility through mass balances for Nitrogen (N), Phosphorous (P), and Potassium (K). The results presented in this study are given as CO_2-equivalents per kg energy corrected milk (CO_2-eq kg^{-1} ECM). The energy correction was calculated with the energy content formula from Sjaunja et al. (1991) and an energy content of 3.17 MJ gross energy per kg ECM. The reference flow is 1 kg ECM of delivered milk, which is defined as sold milk plus private use.

Methane emissions were assessed with a Tier 2 approach. For Norwegian conditions, different algorithms exist that predict the methane emission from ruminant digestion (Storlien et al. 2014). Due to lack of further data, we could only choose between two of these formulas, which are based on dry matter intake and non-detergent fibre intake, respectively. These two algorithms are country-specific and consider feed intake but they do not account for seasonality or detailed diet composition. Hence, they fulfil the requirements for a Tier 2 approach but are not sophisticated enough to be a Tier 3 approach (IPCC 2006, Chapter 10). To avoid introducing a bias caused by the choice of algorithm, we used both formulas for our calculation.

The emission factors for N-inputs (Table 13) as mineral fertilizer (EF1), organic fertilizers (EF2), and crop residues (EF3) are identical and named EF1 in IPCC 2006. The harvested amount was calculated based on the energy demand for milk and meat production and losses after harvest estimated based on Steinshamn et al. (2004).

IPCC gives default values and ranges for both crop residues above ground (harvest losses and stubbles, AGDM) and below ground (roots, BGDM). For grassland on fully cultivated land, we assumed ploughing every 5th year. Thus only 20% of crop residues after harvest became a source for N_2O emissions which is in line the with IPCC 2006 guidelines. Because all farms had grassland and no arable crops, we assume the net carbon sequestration from mineral soils to be zero.

For area classified as peat, we used the default emission factor for N_2O from managed organic soil (IPCC 2006, Chapter 11). CO_2 emissions from these soils were included based on the IPCC Tier 1 methodology for grassland (IPCC 2006, Chapter 6). No farm was located on 100 % peat soil.

Table 13: Description of parameters used in Table 2, their default values and ranges as 95 % confidence interval (CI) as given in source.

Parameter name	Definition/Description	Default value (95 % CI)	Unit	Tier and Source
EF1	N_2O-N emissions as function of mineral N input	0.01 (0.003-0.03)	kg N_2O-N kg^{-1} N	Tier1, IPCC (2006)
EF2	N_2O-N emissions as function of organic N input	0.01 (0.003-0.03)	kg N_2O-N kg^{-1} N	Tier 1, IPCC (2006)
EF3	N_2O-N emissions as function of N in crop residues	0.01 (0.003-0.03)	kg N_2O-N kg^{-1} N	Tier 1, IPCC (2006)
EF4	N_2O-N from droppings during grazing	0.02 (0.006-0.06)	kg N_2O-N kg^{-1} N	Tier 1, IPCC (2006)
EF5	N_2O-N from organic soil	8 (2-16)	kg N_2O-N ha^{-1}	Tier 1, IPCC (2006)
EF6	CO_2-C from peat soil	0.25 (0.025-0.475)	t CO_2-C ha^{-1}	Tier 1, IPCC (2006)
AGDM	Above ground dry matter of crop residues	0.4 (0.2-0.8)	kg DM kg^{-1} yield	Tier 1, IPCC (2006)
BGDM	Below ground dry matter of crop residues	0.8 (0.4-1.2)	kg DM kg^{-1} AGDM	Tier 1, IPCC (2006)
AGDM-N	N content in AGDM	0.025	kg N kg^{-1} AGDM	Tier 1, IPCC (2006)
BGDM-N	N content in BGDM	0.016	Kg N kg^{-1} BGDM	Tier 1, IPCC (2006)

Impact assessment

The impact assessment is only carried out for the climate change impact category. We used the global warming potentials for CO_2, CH_4, and N_2O of 1, 25, and 298, respectively (IDF 2015). Uncertainty of the characterization factors and the actual impacts are assumed to be independent of the farms, i.e. we assume that the impacts from the same amount of methane are the same, irrespective of the methane source.

4.2.3 Monte Carlo analysis

Monte Carlo analysis is a method to quantifying the effects of parameter uncertainties on model outcomes. It has been well introduced into LCA and is frequently used (Heijungs & Huijbregts 2004). During Monte Carlo analysis, values of input parameters are randomly varied according to a specific uncertainty distribution and the effect of each combination on model outcome is observed.

For each of the parameters we generated input distributions that have the same 95 % confidence intervals as the ranges given by IPCC (Table 13). The emissions of N_2O from N inputs are the product of multiple naturally occurring variables such as soil conditions, weather, etc. Hence, the emission factors can be assumed to be lognormal distributed (Vose 1996) and chose the parameters so that the uncertainty ranges given in Table 3 are the 2.5^{th} and 97.5^{th} percentile, respectively. For the emission factor for CO_2 from peat soil (EF6, Table 13) we used a normal distribution with the mean and standard deviation as half the range of the uncertainty as indicated in IPCC (2006). We assumed the parameters for crop residues (AGDM and BGDM) to be normally distributed. Additionally, we used each of the Tier 2 algorithms for the calculation of ruminant methane production (Table 12, Storlien et al. 2014) in half of the calculations.

To assess differences between farms, we compared the median results. The true median is the theoretical median of the model and a given parameter set when we can run infinite Monte Carlo iterations. It was our aim to place the true median in the interval between the 49^{th} percentile and the 51^{st} percentile. To achieve this 10,000 Monte Carlo iterations are required (Morgan & Henrion 1990). This number of simulations was used to assess the effect of changes in the parameters in Table 13 on the carbon footprint of milk production at farm gate on each of the 20 farms. We generated a distribution of the carbon footprints of milk production for each farm and calculated the absolute difference of the median result for each combination of two farms. We tested whether the inter-quartile ranges (IQR) of these distributions show differences between farms with and without peat soil and between organic and non-organic farms using the Student’s t-test in JMP 12.0.1.

Parameter Covariance

When a collection of parameters is varied simultaneously, correlations between these parameters must be accounted for. The best way to account for these correlations is parameter covariance (see e.g. Huijbregts 2003). We have to deal with parameter covariance in two situations. Firstly, we assume complete parameter covariance within each farm for the emission factors EF1, EF2, and EF3 (Table 13). This is because they all concern direct emissions of N_2O after addition of N to the soil and mineral fertilizer, organic fertilizers, and crop residues are influenced by the same soil conditions, crop type, and climate. All other parameters were varied independently of each other.

The second instance of parameter covariance is the interdependency of the assessed farms. Most background processes, e.g. electricity production, fertilizer/feed production or transports, are identical for all farms. As parameters concerning background processes should be varied simultaneously for all product systems (Huijbregts et al. 2001), these variations cancel each other out regarding the direct comparison of farms. Consequently, all uncertainty that is independent of the farmers' management is excluded from the Monte Carlo simulation. Therefore, we leave all parameters constant that are judged covariant and are only varying the parameters given in Table 13.

Comparison indicator

We used the comparison indicator introduced by Huijbregts (1998) to calculate the significance of the absolute differences. The comparison indicator is the quotient of two distributions. In practice that means that for each iteration, the carbon footprint of milk in Farm A (with a higher median carbon footprint) is divided by the carbon footprint of milk in Farm B) with a lower median carbon footprint). The quotient is significantly different from 1 when < 5 % of all iterations are below 1. This means that Farm B has a better environmental performance than Farm A.

4.3 Results

Figure 1 a-d show the generated distributions of GHG emissions per kg ECM for each farm, ordered by organic (a & b) vs. non-organic (c & d) and by peat soil (b & d) vs. no peat soil (a & c). According to the Student's t-test, inter-quartile ranges (IQR) of the distributions are significantly different on farms with peat soil compared to farms without peat soil ($p<0.05$, not shown). The type of farm (organic vs. non-organic management) has no influence on the width of the distribution expressed by IQR.

The uncertainty of results in the single farms, expressed as two-times standard deviation divided by the median result, ranged between 4.2 % (Farm 13) and 15.3 % (Farm 9). This means that 95 % of values in the resulting distribution of one farm are within a range of 4-16 % of the median of that farm. While some farms can clearly be seen as having high or low GHG emissions per kg ECM, all farms have at least some overlapping of results with other farms (Figure 7). This could mean that the uncertainty of the median result of two farms could be misleading when the difference between the farms is not significant.

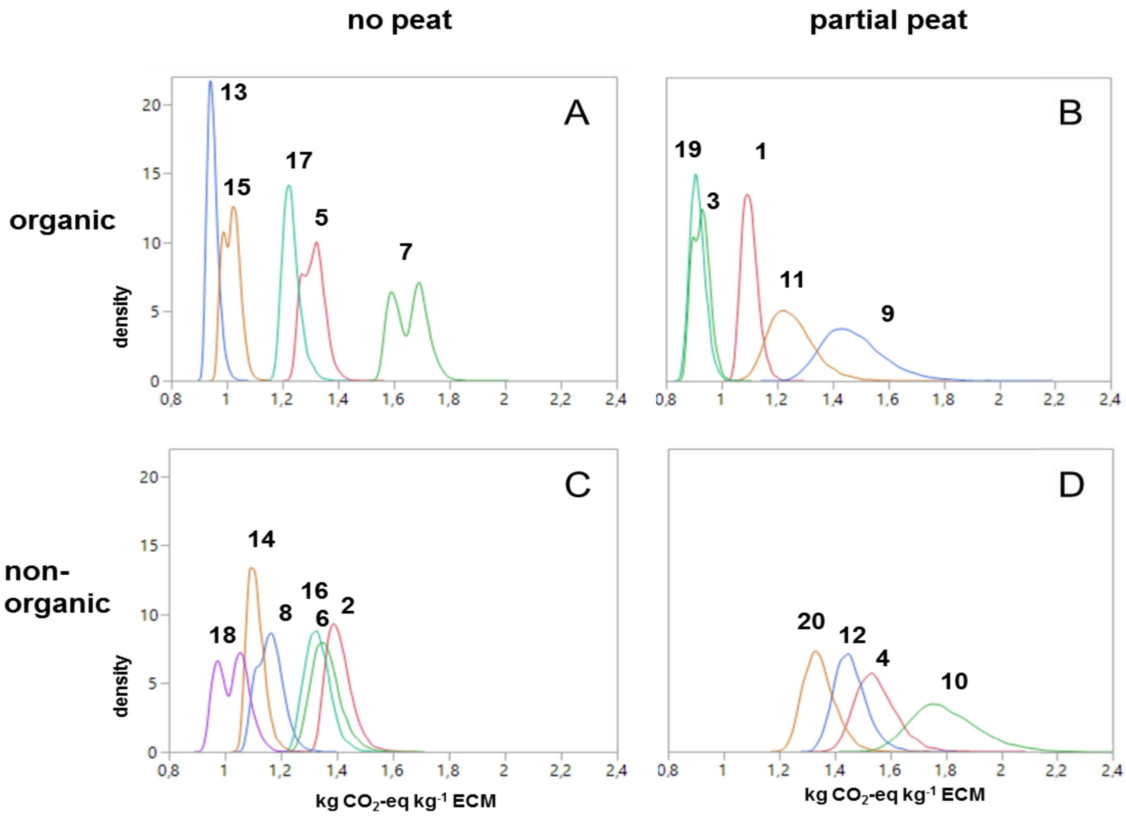

Figure 7: A-D: Distribution of carbon footprint (cradle-to-gate) per kg delivered ECM of analysed farms derived by Monte Carlo simulation. A: organic farms without peat soil, B: organic farms with peat soil areas, C: non-organic farms without peat soil, D: non-organic farms with peat soil areas. Numbers indicate farm ID.

Some farms have two distinct peaks in the distribution of their milk carbon footprint. These peaks are created by the two different algorithms to calculate methane emissions from ruminant digestion. Exemplarily, in the case of farm 18 the right peak, i.e. the one with higher emissions, is the peak from the formula including DM. This is because this farm has very high use of concentrate (55 % of DM intake), which has a lower NDF content compared to roughages. Conversely, the higher peak for farm 7 is caused by the formula using NDF. With 23 % concentrate of total DM intake, the NDF intake is high and consequently a high estimation of CH4 emissions by the NDF formula occurs. The peaks also exist in some other farms but are less distinct. Here and in all other farms the choice of CH_4 algorithms does not seem to introduce a bias.

We calculated the differences between the median results for the carbon footprint of milk in the farms derived by Monte Carlo simulation for each direct comparison between farms (Table 4). The farm with the largest interval around the median between the 49th and 51st percentiles is farm 10 with 0.006 kg CO_2-eq per kg ECM. With a confidence of 95 %, the true median is within ±0.003 kg CO_2-eq of the calculated median (Morgan and Henrion 1990). Therefore, we can safely use 0.01 kg CO_2-eq per kg ECM as precision for the differences between the farms.

The significance of the differences was calculated using the comparison indicator (Table 4). The smallest significant difference is between the farms 13 and 15 with 0.07 kg CO_2-eq ($p<0.05$, highlighted boxes). The largest non-significant difference is 0.20 kg CO_2-eq between farm 7 and farm 9 (highlighted boxes). We can identify the farm with the lowest carbon footprint as farm 19. Here, milk has a median carbon footprint of 0.91 kg CO_2-eq kg^{-1} ECM. However, milk of the farms 3 and 13 has no significantly higher carbon footprint. The highest product-related emissions are from farm 10 with 1.79 kg CO_2-eq kg^{-1} ECM. The differences of farm 10 to all other farms except for farm 7 are significant.

In total, from the 190 direct comparisons, 149 (78 %) differences are significant (Table 4). Above a direct difference of 0.11 kg CO_2-eq kg^{-1} ECM, 95 % of all comparisons were significant. This is 8.7 % of the average of the median results from all 20 farms, and show that the difference between the farms are significant when it is larger than 8.7%.

Table 14: Median results and difference in carbon footprints of milk between the analysed farms in kg CO2-eq per kg delivered milk. Boxes highlight the largest non-significant difference and smallest significant difference (bold), respectively.

Type			con	org	con	org	con	con	con	con	con	org	org	org	con	con	org	con	org	org	org	org
	Farm		10	7	4	9	12	2	6	20	16	5	11	17	8	14	1	18	15	13	3	19
		Median	1.785	1.662	1.542	1.462	1.451	1.402	1.357	1.340	1.328	1.311	1.243	1.227	1.160	1.106	1.099	1.032	1.017	0.947	0.925	0.912
peat	19	0.912	0.87[1]	0.75	0.63	0.55	0.54	0.49	0.44	0.43	0.42	0.40	0.33	0.32	0.25	0.19	0.19	0.12	0.10	0.04	0.01	x
peat	3	0.925	0.86	0.74	0.62	0.54	0.53	0.48	0.43	0.41	0.40	0.39	0.32	0.30	0.23	0.18	0.17	0.11	0.09	0.02	x	n.s.
	13	0.947	0.84	0.71	0.59	0.51	0.50	0.46	0.41	0.39	0.38	0.36	0.30	0.28	0.21	0.16	0.15	0.09	**0.07**	x	n.s.	n.s.
	15	1,017	0.77	0.64	0.52	0.45	0.43	0.39	0.34	0.32	0.31	0.29	0.23	0.21	0.14	0.09	0.08	0.02	x	*	*	**
	18	1,032	0.75	0.63	0.51	0.43	0.42	0.37	0.32	0.31	0.30	0.28	0.21	0.20	0.13	0.07	0.07	x	n.s.	n.s.	**	*
peat	1	1,099	0.69	0.56	0.44	0.36	0.35	0.30	0.26	0.24	0.23	0.21	0.14	0.13	0.06	0.01	x	n.s.	*	***	***	***
	14	1,106	0.68	0.56	0.44	0.36	0.34	0.30	0.25	0.23	0.22	0.20	0.14	0.12	0.05	x	*n.s.*	n.s.	*	***	***	***
	8	1,160	0.63	0.50	0.38	0.30	0.29	0.24	0.20	0.18	0.17	0.15	0.08	0.07	x	n.s.	*n.s.*	*	***	***	***	***
	17	1,227	0.56	0.43	0.31	0.23	0.22	0.17	0.13	0.11	0.10	0.08	0.02	x	n.s.	**	**	**	***	***	***	***
peat	11	1,243	0.54	0.42	0.30	0.22	0.21	0.16	0.11	0.10	0.08	0.07	x	n.s.	n.s.	*	*	**	***	***	***	***
	5	1,311	0.47	0.35	0.23	0.14	0.14	0.09	0.05	0.03	0.02	x	n.s.	*	**	***	***	***	***	***	***	***
	16	1,328	0.46	0.33	0.21	0.13	0.12	0.07	0.03	0.01	x	n.s.	n.s.	*	**	***	***	***	***	***	***	***
peat	20	1,340	0.45	0.32	0.20	0.12	0.11	0.06	0.02	x	n.s.	n.s.	n.s.	*	**	***	***	***	***	***	***	***
	6	1,357	0.43	0.30	0.18	0.11	0.09	0.05	x	n.s.	n.s.	n.s.	n.s.	*	**	***	***	***	***	***	***	***
	2	1,402	0.38	0.26	0.14	0.06	0.05	x	n.s.	n.s.	n.s.	*	n.s.	**	***	***	***	***	***	***	***	***
peat	12	1,451	0.33	0.21	0.09	0.01	x	n.s.	n.s.	n.s.	n.s.	*	*	***	***	***	***	***	***	***	***	***
peat	9	1,462	0.32	0.20	0.08	x	n.s.	n.s.	n.s.	n.s.	n.s.	*	*	**	***	***	***	***	***	***	***	***
peat	4	1,542	0.24	0.12	x	n.s.	n.s.	*	*	*	**	***	**	***	***	***	***	***	***	***	***	***
	7	1,662	0.12	x	n.s.	n.s.	*	***	***	**	***	***	***	***	***	***	***	***	***	***	***	***
peat	10	1,785	x	n.s.	*	*	**	***	***	***	***	***	***	***	***	***	***	***	***	***	***	***

[1] Median in row subtracted from median in column, corresponding significance levels in lower part of the table. Significance of difference calculated using comparative indicator (see text). * less than 5% of values below 1, ** less than 1% of values below 1, *** less than 0.1% of values below 1, n.s. not significant. Non-significant differences additionally highlighted yellow.

4.4 Discussion

4.4.1 Benchmarking the carbon footprint of milk from practical dairy farms

In this study, we examined the effect of parameter uncertainty suggested by IPCC (2006) on the ability to differentiate between different farms. With an existing dataset of dairy farms we focussed on direct and indirect N_2O-emissions and on CO_2-emissions from soils. These are frequently estimated with IPCC Tier 1 approaches and might be appropriate in practical surveys. We did not examine overall uncertainty of results, which might be caused by uncertainty in the supply chain, unspecific data sets, or variations in GHG emissions outside the range suggested by IPCC. We rather wanted to look at how uncertainty generally affects the comparability of carbon footprints of milk produced on different farms within the same framework.

Through direct comparison of the medians, we can clearly identify farms with highest and lowest product-related GHG emissions. From the 190 direct comparisons in Table 4, 149 (78 %) differences are significant. A significant difference between two farms means that the farm with higher emissions performs worse in regard to carbon footprint of milk compared to the other one. The difference between these two farms is therefore larger than the uncertainty in GHG emissions from managed soils calculated with IPCC Tier 1 and caused by N_2O and CO_2 emissions. We think this demonstrates that IPCC Tier 1 methodology to estimate GHG emissions of soil management can be suited to judge the environmental performance of milk production on farms concerning the carbon footprint. This is valid for farms within one study using the same modelling choices when the uncertainty in GHG emissions from managed soils is not larger than suggested by IPCC (2006).

For a majority of direct comparisons, we can accept our hypothesis that the variability of the calculated carbon footprint of milk between farms is higher than the suggested effect on the results caused by uncertainty of emission factors given by IPCC. Consequently, despite the use of IPCC Tier 1, we can differentiate the carbon footprints of milk from different farms. This allows us to analyse management factors that increase or decrease the carbon footprint - here of milk production - and guide future optimisation processes to mitigate GHG loads for farmers using whole-farm models.

4.4.2 Embedding results in current scientific knowledge

In an ex-post Monte-Carlo analysis of emission factor uncertainty on 47 French dairy farms Chen and Corson (2014) found an uncertainty of 2-7 % for global warming potential which is lower than the range of 8.7 % we have found in 20 Norwegian dairy farms. The parameter set in that study is somewhat different, as Chen and Corson included uncertainty from ammonia emissions and uncertainty from the algorithm for CH_4 from ruminant digestion itself. In our study, we looked only at uncertainty of N_2O emissions from fertilization and crop residues as well as of CO_2-emissions from peat soil areas but under inclusion of uncertainty from choice of CH_4 algorithm.

Zehetmeier et al. (2014) analysed the effect of modelling choices on variability and epistemic uncertainty. For a system, where allocation is performed by system expansion and credit for meat production from a suckler cow system, uncertainty was as high as 93 % due to lack of knowledge about the beef producing system. Uncertainty just from the dairy farm (before any allocation or credit) was found between 8-10 %. Basset-Mens et al. (2009) analysed uncertainty from emission factors. They found an uncertainty of 7 % of the carbon footprint of milk in regard to emission factor uncertainty which is also comparable to the uncertainties found in this study. In summary, recent analyses using Monte Carlo simulation found result uncertainties of dairy farming very similar to our results.

4.4.3 Methodological considerations

In our study, we made the assertion that uncertainty from all background processes can be judged as covariant. However, this is only true when the background processes are in fact identical. One major concern about this assumption could be the fact that the background processes for agricultural processes (i.e. concentrate production) are different depending on whether the farm is organic or non-organic. For organic farms, all agricultural processes in the supply chain also have to be organic. If it is assumed that organic crop production as a background process has not the same uncertainty compared to non-organic production, then in our setting organic farms could only be compared to other organic farms.

On the other hand, we are currently not aware of findings of generalized significant differences in GHG emissions on the regional and crop specific level between organic and non-organic production. While differences in the carbon footprints between organic and non-organic production exist under comparable soil and climate conditions (e.g. Tuomistio et al. 2012, Skinner et al. 2014, Lee et al. 2015), these do not seem to be generalizable across multiple crops and regions. Therefore, we have to regard the uncertainty associated with the underlying GHG emissions as independent of the farmers' management concerning organic or non-organic, which in our study leads to the assumption of covariance.

For grassland on organic soil, the IPCC guidelines (IPCC 2006, Chapter 6) suggest an emission factor of 0.25 tonnes CO_2-C ha^{-1} yr_{-}^{1} with a range of 0.025-0.475 tonnes CO_2-C ha^{-1} yr^{-1}. This means CO_2-emissions from 92-1742 kg CO2 ha^{-1} yr^{-1}. Measurements in Germany suggest that emissions from peat soils on grassland could be as high as 5000 kg CO_2-C ha^{-1} $yr^{-}1$ (Tiemeyer et al. 2016). We ran the same simulation including this high assumption. This led to a shift of all farms with peat soil to a higher carbon footprint. We then performed the same comparison based on the comparison indicator and found that only 21 direct comparisons (11 %) remain non-significant. The real uncertainty might be different from the uncertainty given in IPCC and it might have an important impact on results. Consequently, IPCC Tier 1 can only be used for comparisons when the uncertainty range given in IPCC matches the real uncertainty of the analysed processes.

The IDF guidelines for carbon footprints in dairy farming require the use of Tier 2 methodology throughout. We have shown, however, that the use of Tier 1 methodology for the estimation of N_2O and CO_2-emissions from soils is acceptable given the differences between farms are high enough (here 8.7 % of the median carbon footprint of milk from 20 farms). We therefore encourage the authors of the IDF guidelines to allow for the use of Tier 1 methodology. As the guidelines already demand an uncertainty assessment, the necessity to use at least Tier 2 can be drawn from the results of the Tier 1 analysis. This procedure underlines the iterative nature of full or partial life cycle assessments. The procedure as described in this study could be used as a framework.

Finally, the IDF guidelines should specify the data quality demanded for processes in the supply chain, especially in regard to imported feed. Further research should be directed in the analysis of covariance of carbon footprints of feed imports and whether our assumption of covariance can be maintained or should be rejected. Additionally, this supports the need for documented supply chains, e.g., in future certification processes of environmental performance of milk production.

4.5 Conclusions

The hypothesis for this study was that a significant differentiation of the milk carbon footprint between farms is possible with a Tier 1 approach for soil emissions of N_2O and CO_2. Methodologically this is the case when the variation of the carbon footprint of milk between farms is higher than the uncertainty of the calculation results.

We were able to accept this hypothesis for a majority of our comparisons and found that a difference above 8.7 % of the population median is sufficient to establish significance.

However, some conditions must be met for our results to be generalized. It must be established that background processes, especially in the use of datasets for import feed, can be judged covariant in order to prohibit them to influence the comparison between farms. Secondly, the uncertainty ranges given in IPCC Tier 1 must be appropriate for the assessed systems.

In regard to the further development of the IDF guidelines for carbon footprints in the dairy sector, the data quality demands for data concerning the supply chain of dairy farming should be clarified. The use of IPCC Tier 1 methodology should be allowed, given an uncertainty assessment in regard to its use. Furthermore, guidance for the procedure of uncertainty assessments and the interpretation thereof should be given explicitly.

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5 General discussion

The discussion consists of three sections. Firstly, generalizable findings from the articles in Chapters 2, 3, and 4 are presented and the demand for both qualitive and quantitive uncertainty analysis is shown. In the second section, limitations of carbon footprints of milk production are discussed qualitively. The third section proposes how an implementation of quantitative uncertainty analysis could be applied on carbon footprints in dairy farming. In order to ensure correct meaning in the following the definitions of activity data and emission factor from the IPCC (2006) are given below.

Activity data: Data on the magnitude of a human activity resulting in emissions or removals taking place during a given period of time. […]

Emission factor: A coefficient that quantifies the emissions or removals of a gas per unit activity. Emission factors are often based on a sample of measurement data, averaged to develop a representative rate of emission for a given activity level under a given set of operating conditions.

5.1 Main findings from the articles

In the previous sections, different methodological considerations of dairy carbon footprinting were presented. Each of these chapters has its special topic and its own discussion dealing with these topics. However, some of the findings were mentioned repeatedly or are generalizable and lead to overarching conclusions that could not be drawn from one single analysis.

Farm activity data is variable.

Data representativeness is a major concern for the interpretation of results in life cycle assessment (ISO 2008a). In order to judge representativeness, variability of activity data has to be accounted for. In agriculture both inter-annual variation and intra-annual variation occur. Management can change between years (inter-annually) to adapt to different crop yields, whether conditions, or animal-based variation. The extent of this variation has been analyzed in Chapter 3. Intra-annual variation means that agricultural management can change within a year, e.g., when choosing pastures during the summer months as seen in Chapters 3 and 4 or by strategically selling (or keeping) animals.

Furthermore, variability also occurs on a smaller level in dairy farming. While using herd average milk yields for the calculation of the feed intake (Chapter 3 and 4), each animal has a different energy demand based on its specific milk yield, weight gain, and metabolism. The effect of this intra-herd variability on the milk carbon footprint has not yet been analyzed. Similarly, feed quality may be different for yields from different fields or even change during storage and thus affect the ruminant methane production of the animals. In toto, farming makes use of many natural processes, be it on microbiological, macrobiological or meteorological level. These processes vary naturally within a broad spectrum, making it difficult to obtain representative data.

Farm activity data is uncertain.

Uncertainty of farm activity data is driven by two causes. Firstly, not all necessary data can be measured under practical conditions. This concerns losses during feeding, gaseous and liquid losses during storage of feed, changes of feed quality, etc. A second driver for activity data uncertainty appears when the variability of processes is higher than sample density. For example, frequent analyses of feed quality were performed on the research farm Trenthorst (Chapter 3) and for the grass silage on the Norwegian dairy farms (Chapter 4). However, feed quality may change within a silage stack but also after opening through secondary fermentation (Kamphues 2014). Consequently, despite best efforts, some data that is needed for calculation remains uncertain. Typically, important uncertainties are crop yields and qualities including water content, losses and changes of quality during feed storage, losses during feeding, and amount of produced milk and milk losses based on monthly milk recordings.

Main emissions must be calculated based on uncertain emission factors.

Main drivers of the carbon footprint in dairy farming are direct emissions of N_2O during crop production and direct emissions of CH_4 from ruminant digestion as well as emissions from handling and storage of animal manure (Chapter 3, figure 3). In order to measure N_2O emissions during crop production, static or dynamic chambers or eddy covariance are typically used (Freibauer & Kaltschmitt 2003, Henseler & Dechow 2014). As crop fields often have a natural variation of soil properties, variability of N_2O emissions within fields occur which has to be accounted for by using enough measuring devices. This makes the process of measuring N_2O emissions during crop production expensive and prohibits the permanent measurement under practical conditions.

Methane emissions from ruminant digestion can be measured in dedicated climate chambers (Derno et al. 2005) or by tracer gas methods (Deighton et al. 2014) where each cow carries a measuring device. Both of these methods are not transferable to everyday conditions. Some experimental sites aim to measure CH_4 emissions in open stables by measurements of concentration and wind profiles (Winter & Linke 2017). However, so far these also do not provide transferable results and do not work on pastures.

As a consequence, with N_2O and CH_4, the main emissions during milk production have to be estimated using emission factors. These are based on physical relationships of emission data with measureable data of activities that act as a driver for the emission. For dairy carbon footprints according to the IDF guidelines, these emission factors follow the IPCC Tier system of increasingly complex emission prediction models (IDF 2015, Chapter 4). In general, a higher IPCC Tier reduces uncertainty of the emission factor but more detailed activity data is needed. In order to estimate the uncertainty of emissions, both the uncertainty of the emission factor and the uncertainty of the activity data are needed.

A thorough quantitative analysis of variability and uncertainty is necessary.

The IDF guide for carbon footprints in the dairy sector (IDF 2015) suggests to estimate the uncertainty of data based on either quantitative approaches or qualitatively through discussion (IDF 2015, p. 25). From the analyses in Chapters 3 and 4, as well as the discussion until this point, it should become clear that many sources of uncertainty exist and these uncertainties interact with each other. Interaction means that uncertainties do not simply add up but are restricted within the flow model by mass, nutrients, and energy balances. In order to obtain a thorough understanding of the system, allow fair comparisons between farms (farming systems, regions, products, etc.) and to allow the identification of improvement measures and their success probabilities, a qualitative discussion of uncertainty is not enough and additionally a quantitative uncertainty assessment should be performed.

When dairy LCA uncertainties are discussed only qualitatively, target audiences may lack trust in conclusions drawn from the results. On the other hand, it is not difficult to calculate the uncertainties of different scenarios, overall uncertainties, and significance of differences. By clearly communicating quantified uncertainties, the usability of LCA results can be improved and confidence can be increased especially when using LCA for strategic or operational decisions.

5.2 Qualitative discussion of model uncertainty

Before a quantitative uncertainty assessment can be performed, a qualitative discussion of uncertainty is necessary. The main question for any uncertainty discussion is, in how far results and conclusion are affected by uncertainty. For carbon footprints of milk production that aim to compare different farms, two limitations in regard to LCA according to ISO 14040/14044 are obvious. Firstly, the limitation from cradle-to-gate instead of cradle-to-grave and secondly, the limitation to global warming instead of a “comprehensive set of environmental issues related to the product system being studied” (ISO 2006b, 4.4.2.2.1 (4)).

5.2.1 Limitation cradle-to-gate

Definition of functional unit

The ISO norms for LCA (ISO 14040 a/b) focus on phases in a product’s life cycle. It is explicitly stated that “[t]he essential property of a product system is characterized by its function and cannot be defined solely in terms of the final product” (ISO 2008a). The variation in fat and protein content of milk at the farm gate is dealt with through the use of ECM. Other variations of the chemical composition are ignored. Furthermore, other (financially) valued aspects of the quality of milk, such as “produced in accordance with guidelines for organic agriculture”, “produced with cows’ access to pasture”, etc. are not taken into consideration of the function that is fulfilled by the product. Consequently, we must question whether milk from different production systems is functionally equivalent in regard to LCA.

In order to provide equivalency, with ECM a “quality corrected” product unit is used (Schau & Fet 2008). Suggestions to include economic elements are frequently made, as well as the inclusion of intensity or land use into the functional unit, e.g. “1 kg ECM from extensive grassland“ (see e.g. Guggenberger & Herndl 2017, Casey & Holden 2006, van der Werf et al. 2009). From a purist point of view, different farming strategies cannot be compared when the farming strategy itself is included into the functional unit (i.e. “milk from organic farming” vs. “milk from non-organic farming”), as they do not provide the same function. An alternative concept to assess land use, including the quality of the used land, is the hemeroby concept (see e.g. Klöpffer & Grahl 2014, Brentrup et al. 2002).

In conclusion, the use of ECM as functional unit is not entirely in line with ISO methodology and aims. For any dairy carbon footprint, the equivalency of the compared units must be assured or the conditions under which equivalency can be assumed must be named.

5.2.2 Limitation carbon footprint

The IDF guidelines (IDF 2015) concern only carbon footprints, i.e. only the global warming potential associated with the production of milk. Other important environmental issues such as eutrophication and acidification potentials and abiotic and fossil resource use are not part of studies following the IDF guidelines. Hence, any analysis of a farm or any comparison between two farms or farming systems remains incomplete. Furthermore, agriculture contributes to 4-18 % of all GHG emissions worldwide. Other environmental issues such as eutrophication, land use and land use change, soil degradation are mainly driven by agriculture. For example 94% of all NH_3

emissions, which are relevant for eutrophication and acidification, in Germany are caused by agriculture (Umweltbundesamt 2014).

While it has been shown that the GWP of non-organic milk production is similar to the GWP of organic milk production (Hülsbergen & Rahmann, 2015), the nitrogen intensity and product related eutrophication potentials of organic agriculture are lower compared to non-organic production (Koesling, 2017). Consequently, limiting the environmental assessment to GHG emissions may not adequately reflect differences in the environmental performance of different farming systems.

5.3 Comparing milk carbon footprints from practical dairy farms

As shown so far, the uncertainties from variability, emission factors, and lack of knowledge under practical conditions raise the demand for quantification and interpretation of uncertainty when dealing with carbon footprints in the dairy sector. The tools needed for this quantification are well developed and published and not overly complex.

Hence, in the following section some existing knowledge for uncertainty assessments of dairy carbon footprints is discussed. It consists of three sections. Firstly, the underlying principles of quantitative risk assessment are presented. Secondly, the procedure in the ideal situation of comparing two farms or two farming systems within the same study is described. It is a quantitative approach of error propagation with a Monte Carlo simulation based on quantitative risk assessment methodology. It can be used when the environmental assessment is performed using a flow model. As the last section, it is described how one could use uncertainty information when interpreting results from two separate studies.

5.3.1 Concept of variability and uncertainty

Uncertainty assessment is required according to ISO 14044 (ISO 2008b) when intending to publish results for comparative assertions. Uncertainty is defined as incomplete knowledge (Haimes 1998). This can be reduced by gaining more knowledge, e.g. by measuring more data points or measuring more accurately. Variability, on the other hand, is the stochastic change of a quantity (ibid.). This does not mean that this quantity is uncertain but a prediction of a future value of this quantity is impossible. Under normal circumstances, ex-post analyses do not require prediction. However, it may be impossible to accurately describe a system when representative quantities (e.g. crop yields) cannot be determined from existing data (see Chapter 3). Then the stochastic variability of a quantity leads to incompleteness of knowledge – which is uncertainty. As a result, both variability and uncertainty prevent from knowing the true value of a quantity which is either knowledge uncertainty or variability uncertainty (Haimes 1998). Haimes argues that they can be used without a need to distinguish, except when analyzing the difference between the two.

In Chapter 4, Monte Carlo Simulation was introduced and applied in the agricultural context. The approach taken in Chapter 4 was an ex-post uncertainty analysis where first the model was built and the uncertainty analysis was added to the model later. The approach led to a reduction of possibilities in the Monte Carlo simulation, partly restricted by the available features of the used software.

Naturally in order to circumvent such restrictions, all demands for Monte Carlo simulation must be known in advance. From Chapter 4 the following very general demands arise for using Monte Carlo simulation:

1. Ability to use variables instead of scalar values in the flow model. These variables (or parameters) are varied in the Monte Carlo simulation.
2. Ability to generate distributions for the variables according to type of distribution (normal, lognormal, triangular, etc.) to use as input for Monte Carlo simulation.
3. Ability to relate distributions towards each other in order to force (or avoid) co-variance of parameters.
4. Ability to use distributions in the calculation.

The need for addressing correlation of input parameters is shown in Huijbregts (2003). The effect of correlated input parameters on LCA results is demonstrated in Bojaca & Schrevens (2010) and discussed in Chapter 4 of this thesis. However, to this date the most commont LCA software (GaBi, OpenLCA, SimaPro, Umberto) do not support parameter correlation. In their respective Monte Carlo Simulations, only independent variables are supported.

5.3.2 Execution of uncertainty assessment

A scenario is a collection of parameter values and their associated probability distributions. It can be activity data for a single farm in a single year, average activity data for a representative farm in a region, or any other collection of parameters to be compared with a different collection.

Once the flow model has been established, a probability distribution must be defined and generated for each uncertain parameter (see e.g. Table 13, Chapter 4). The probability distribution can be derived from literature, databases (e.g. the ecoinvent LCI database), or estimation. When the shape of the distribution is unknown, it is best practice to define a minimum value, a maximum value, and a most-likely value and to use a triangular distribution (Vose 1996).

In the next step, the relationships (dependencies) between these distribution must be established. This is typically done via rank order correlation by defining a Spearman's rank order correlation coefficient (Vose 1996). Since more than two distributions will have to be related to each other, a correlation matrix must be established, in which the correlation coefficient for each combination of two parameters is set. This includes interdependencies of the parameters between scenarios (Huijbregts et al. 2001), or in the case of dairy carbon footprints: farms or farming systems. This matrix must contain all parameters to be used in the study, that means that when a comparison between two or more scenarios (different farms, farming systems, etc.) should be performed, each parameter must be present for each farm. The mathematical demand for such a matrix is that it is positively determined, i.e. no negative eigenvalues may exist (Vose 1996). As a last preparatory step, the relationships defined in the correlation matrix are modelled on the parameter distributions.

In the next step, the Monte Carlo simulation is executed by either iterating the entire model calculation over the parameter distributions as done e.g. in Umberto 5 or by iterating each calculation step over the entire distribution as done e.g. in the python package mcerp (Lee 2018).

As a result we obtain distributions for all output parameters. These can be the result of the assessment (e.g. kg CO_2-eq per kg ECM) but it should generally be possible to additionally output intermediate calculation steps, i.e. the distribution of LCI data. To compare distributions from different farms or different scenarios within the same study the comparison indicator was introduced by Huijbregts (1998).

5.3.3 Comparison of separate studies

In order to compare results from separate studies it must be ensured that goal and scope as well as underlying modelling principles are equivalent. A realistic scenario could be that two studies were performed according to the guidelines of the IDF but include a quantitative uncertainty assessment. The result would be communicated in mean result and uncertainty (as two times the standard deviation of the mean).

In this thesis it is suggested for this scenario to use again the comparison indicator to judge whether two carbon footprints are different from each other. While other methods exist, such as the computation of the overlap of two normal distributions (Inman & Bradley, 1989), the comparison indicator can easily be calculated using spreadsheets or programming languages and can be interpreted directly.

As an example, the farms 1, 4, 9, 11, and 12 from the study Schuler et al. (2018a, Chapter 4) were again compared using the median result and the standard deviation as input data for a Monte Carlo simulation in MS Excel®. The distributions were modelled using 10.000 lines of the formula

```
Cell_N = NORM.INV(RAND(), mean, sdm)
```

with `mean` as the mean from the distribution of the carbon footprint of the farm and `sdm` as the standard deviation of the carbon footprint. The farms are significantly different when more than 9500 times one farm has a lower result than the other farm.

Table 15: Comparison indicator for selected farm comparisons based on data in Schueler et al. (2018a)

Farm	1	4	9	11	12
1	x	0	3	415	0
4	***	x	7126	9952	8384
9	***	n.s.	x	9434	5561
11	*	**	n.s.	x	228
12	***	n.s.	n.s.	*	x

Comparing each possible combination shows that all significance levels are identical to those in Chapter 4, except for the comparison of farm 9 with farm 11, where the comparison indicator misses significance closely. The similarity is due to the fact that the uncertainty used for the comparison does not comprise the entire uncertainty of each farm's result but only the uncertainty towards the Tier 1 approach for the estimation of GHG emissions from managed soils as described in Chapter 4.

Consequently, this simplified approach is a robust and easy to use method to compare whether results from different studies are significantly different from each other. Using this approach on two separate studies assumes that all variables are independent from each other and does not consider covariance of parameters across studies such as uncertainty from same feed supplies or electricity mixes. The method tends towards judging two farms as not significantly different from each other compared to an analysis where covariance is considered and can therefore be assumed to be a conservative approach.

6 Conclusion

The aim of this dissertation was to challenge the existing common framework for carbon footprints in the dairy sector. In three separate scientific papers it was shown that existing rules on the use of the functional unit, the temporal scope, and demands on data specificity due to uncertainty may not be precise enough to allow for comparison of farms or farming systems when using the IDF guidelines, or are too strict and thereby hindering the execution of carbon footprint studies.

For the functional unit the following mandatory definition was proposed: “The functional unit is 1 kg energy corrected milk (ECM) at the farm gate (including private use, if applicable). The energy correction is performed using the formula given by IDF (2015) and scales to 3.17 MJ per kg ECM.”

For the temporal scope it was shown that variability of crop production prohibits from using production data from a single year. When only a few years are used to calculate the average performance, a sensitivity analysis should be performed by choosing different values for the average yield and observe the effect on the results. An overlapping of bandwidths of adjacent time periods means that the true environmental performance of the studied farm is included in the temporal scope, and is therefore representative with regard to inter-annual variation of crop yields.

In an extended uncertainty assessment using the comparison indicator, it was shown that – if certain conditions of parameter dependency are met – rules of the IDF could be loosened without compromising the ability to compare results between farms. In fact, uncertainty from a bad definition of ECM has been found to be larger than uncertainties from Tier 1 emission factors. Consequently, as a result of this thesis, the IDF guidelines for carbon footprints in the dairy sector should be revised.

Another outcome of this thesis are methodological considerations of uncertainty assessments in LCA for agriculture. It was described and exemplarily executed how to deal with uncertainty of input data and uncertainty of emission factors. Furthermore, recommendations were given on how to interpret LCA results with information on uncertainty and how to use these information to compare different dairy farms and potentially find favorable farming practices.

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